ANALYSIS OF PROPOSED REVISIONS TO CANADIAN LIGHT DUTY MOTOR VEHICLE EMISSIONS STANDARDS

BY

Pilorusso Research Associates Incorporated

FOR

Environmental Protection Service
Environment Canada

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Prepared for:
ENVIRONMENT CANADA

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At the time of writing, the scientific data base required for a rigorous analysis of the benefits and costs of implementing more stringent light duty vehicle emissions standards was far from complete. The analysis required sifting through a great deal of evidence, determining what was significant, and reporting it. Determining which issues are significant and the amount of emphasis that should be placed on each is, of course, a matter of judgement. Where it was necessary to make such judgements Environment Canada staff officers were consulted. In this regard we wish to acknowledge the contributions of V. Shantora and P. H. Giles of the Environmental Protection Service (EPS).

J. A. Korwin of the Technical Advisory Group on Impact assessment of the Treasury Board of Canada Secretariat provided the guidance necessary to ensure that this analysis meets requirements for Socio-Economic Impact Analyses. On some of the more complex issues such as defining "allocative costs" and choosing the appropriate base year for calculating net present values, we have relied on his recommendations.

The analysis of employment and output effects reported in section 4.6.5 was prepared by Dr. P. A. Victor of Victor and Burrell Research and Consulting. Dr. Victor also assisted in the analysis of the nonallocative effects that are covered in this report.

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SUMMARY

This report contains an analysis of the costs and benefits of revising Canada's light-duty vehicle emission standards. The current standards of 1.24 hydrocarbon (HC), 15.5 carbon monoxide (CO), and 1.92 nitrogen oxides (NOx) grams/kilometre (2.0 HC, 25 CO, 3.1 NOx g/mile) came into effect in 1975. The light-duty vehicle emission standards under consideration of 0.25 HC, 4.35 CO, and 0.62 NOx g/km (0.42 HC, 7.0 CO, 1.0 NOx g/mile) are substantially identical to those that have been in effect in 49 of the U.S. states (all except California) since 1981. Permissible levels of emissions from new motor vehicles sold in Canada are established by Transport Canada under the authority of the Motor Vehicle Safety Act.

The analysis was carried out in accordance with Chapter 490, Appendix E, of Treasury Board Canada's Administrative Policy Manual.

The major sources of data and background information for the analysis are: consultants' reports that were commissioned by Environment Canada, in-house studies prepared by Environment Canada staff, and the open literature.

Contribution of Light-Duty Vehicles to Air Pollution

The exhaust of gasoline-powered vehicles is a source of three primary air pollutants: HC, CO, and NOx. Hydrocarbons are also released to the atmosphere through evaporation from the fuel tank and carburetor and during refueling. Carbon monoxide and NOx are believed to have direct effects on health, vegetation, and materials. Hydrocarbons mainly cause adverse effects indirectly through their contribution to the formation of photochemical oxidants. Ozone and other photochemical oxidants are produced by the action of sunlight on mixtures of HC and NOx. Nitrogen oxides emissions contribute to the acidification of precipitation.

Light-duty vehicle emissions are the largest single source of HC, CO, and NOx emissions in Canada. In 1980 light-duty vehicles emitted 460 000 tonnes of

HC, or 23 percent of total HC emissions; 4 719 000 tonnes of CO, or 45 percent of total CO emissions; and 356 000 tonnes of NOx, or 20 percent of total NOx emissions.

Adoption of the proposed emission standards would result in reductions in light-duty vehicle emissions. By 1995 the reductions in light-duty vehicle emissions would be as follows: HC emissions would decrease by 122 000 tonnes per year or 47 percent; CO emissions would decrease by 2 211 000 tonnes per year or 52 percent; NO_x emissions would decrease by 161 000 tonnes per year or 45 percent.

Costs to Meet the Proposed Emission Standards

Approximately 15 percent of the 1984 model year cars sold in Canada will be fitted with emission control systems that meet the proposed emission standards. Assuming that this percentage does not change over the next few years, adoption of the proposed emission standards would result in 85 percent of the passenger cars sold being equipped with more expensive emission control hardware than would be the case if the current standards are maintained.

For passenger cars that require upgraded emission control systems, the maximum additional cost at the vehicle manufacturer level is estimated to average $143 per car. The average retail price equivalent per car of the cost of the additional emission control hardware is estimated to be $235 per car. Both estimates are in 1983 Canadian dollars and are based upon the technology and costs in 1981 when these emission control systems were first used in the United States. These costs may be somewhat lower by the time that the proposed emission standards may be implemented due to advances in technology, reduction in manufacturing costs as the manufacturers gain more experience with the systems, and a drop in the cost of electronics.

During the first ten years that the proposed standards are in force, the total cost to the manufacturers would be 98 to 121 million constant 1983 dollars per year depending upon sales in any given year, or about $143 per car. The maximum cost to the consumer would be 163 to 202 million constant 1983 dollars, or about $235 per car.

The retail price equivalent is an estimate based upon the manufacturers' cost plus average industry markups. The actual cost to consumers will depend upon a number of factors. In the short term, individual manufacturers will set retail prices based upon the degree of price sensitivity of the market, the amount of price competition, how well their products are selling relative to competitors, and other factors. Therefore, in the short term, retail prices may bear little relationship to the manufacturers' costs. However, implementing the proposed emission control standards do incur real costs and ultimately the consumer can be expected to bear those costs.

If the retail price of passenger cars increased by the estimated retail price equivalent, the increase would represent a 2.2 percent increase over the average retail price of passenger cars sold in Canada during 1983.

Adoption of the proposed emission standards would result in all light-duty vehicles sold in Canada being equipped with catalyst emission control systems that require unleaded fuel for proper operation. Currently, about 20 percent of new passenger cars can operate on leaded fuel and 80 percent require unleaded fuel. As a result, about ten years after the proposed standards are implemented the grade split for leaded/unleaded gasoline for road vehicles would shift from 30/70 (i.e. the estimated split if the existing standards are maintained) to 10/90. The Petroleum Association for Conservation of the Canadian Environment has estimated that up to $560 million in refinery capital costs and up to $80 million in annual operating costs would be required to produce the additional unleaded gasoline and to discontinue use of methylcyclopentadienyl manganese tricarbonyl (MMT) in unleaded gasoline. Both estimates are in current (1984) dollars.

Estimates, prepared by Monenco (1984) for Environment Canada, of the costs to produce the additional unleaded gasoline at current market octanes are $494 million in refinery capital costs and $42 million in annual operating costs if MMT is not used in unleaded gasoline. If MMT is used, Monenco estimates that refinery capital costs will be $68 million and annual operating costs $19 million. The Monenco estimates are in current (1984) dollars.

Benefits of Reduced Motor Vehicle Emissions

The near-term benefits of reduced light-duty vehicle emission levels that may be expected relative to the situation that would prevail if emission standards are maintained at current levels are:

- a reduction in the incidence of adverse health effects due to exposure to elevated levels of carbon monoxide (CO), nitrogen dioxide (NO_2), and ozone (O_3).

- Increased production of forests and agriculture crops in areas where production is reduced due to ambient ozone concentrations.

- A decrease in the incidence and severity of short-term acidification of freshwater during spring melt of snowpack containing nitrate.

- The reduction in the use of leaded gasoline will result in reductions in emissions of lead, ethylene dibromide (EDB), and ethylene dichloride (EDC) and a concomitant decrease in the risk of adverse health effects due to exposure to ambient lead, EDB, and EDC.

There are other potentially significant long-term benefits of reductions of HC, NOx, and CO emissions, and the resulting reductions in ozone concentrations and the amount of nitric acid in acid rain. There is a risk of adverse effects beyond those that can be currently identified.

Lowering the level of light-duty vehicle emissions will reduce the probability of occurrence of these adverse effects.

The use of unleaded gasoline results in reduced engine and exhaust system maintenance requirements and improved fuel economy. It is estimated that consumers will save about $1 per 1 000 kilometres driven if they switch from leaded to unleaded gasoline. Based on the new car fleet in 1982 the average fuel economy improvement for passenger cars operating on unleaded fuel has been estimated to be 2.8 percent.

Summary of Costs and Benefits

The costs and benefits of implementing the proposed light-duty vehicle emission standards are summarized in Table S.1. All costs and benefits that can be expressed in monetary terms have been discounted at 10 percent per year to 1983. The dollar figures are in constant 1983 dollars.

Table S.1

Summary of Benefits and Costs Of Implementing The Proposed Light-Duty Vehicle Emission Standards (1987 - 1996)

			Value * (millions 1983 dollars)
Benefits			
1.	Reduction in the incidence of adverse health effects from exposure to ambient concentrations of CO, NO_2, O_3, and lead.		**
2.	Increased production of forests and agricultural crops in areas where reduced production due to ozone concentrations is observed.		**
3.	A decrease in the contribution of nitrates to precipitation acidity.		**
4.	Intangible and unknown benefits of reduced HC, CO, and NOx emissions.		**
5.	Improved fuel economy due to increased use of unleaded fuel.		75
6.	Lower maintenance costs due to increased use of unleaded fuel.		68
Costs			
1.	Cost of emission control equipment		
	(a) Allocative costs		
	Cost to manufacturer	538	
	(b) Nonallocative costs		
	Vehicle manufacturer markup	124	
	Federal sales tax	59	
	Dealer markup	124	
	Research and development	51	
	Subtotal	358	
	Total cost to the consumer		896
2.	Cost to produce additional unleaded gasoline		
	(a) Refinery capital and operating costs		76 - 459***
	(b) Markups and taxes on the increased cost to refiners		**

* Net present value in 1983 at a discount rate of 10 percent.

** Dollar values could not be established.

*** The estimate of $76 million does not include the costs to remove MMT from unleaded gasoline. The estimate of $459 million is the higher of the two available estimates that include costs to remove MMT.

The impact of the increased cost to refiners to produce additional unleaded gasoline would be as follows. The retail price of unleaded gasoline is, on average, $0.024 per litre higher than leaded gasoline. Switching from leaded to unleaded gasoline will cost the consumer an additional $33.82 per year for a car that is driven 16 000 km per year and achieves 11.7 km per litre. This additional cost would be completely offset by a saving of an estimated $33.60 per car per year due to lower maintenance costs and better fuel efficiency.

If the use of MMT in unleaded gasoline is discontinued the retail price equivalent of the increased cost to the refiners is between one third and one half cent per litre, depending on the province in which the gasoline is sold.

Cost-Effectiveness Comparisons

Assuming that the costs to reduce light-duty vehicle emissions can be equally divided amongst control of HC, CO, and NO_x, the approximate cost, at the retail level, per tonne in constant 1983 dollars for emission reductions are: $432/tonne of HC, $26/tonne of CO, and $355/tonne of NO_x. If only the allocative costs of the emission control hardware are considered, the figures become: $240/tonne of HC, $14/tonne of CO, and $198/tonne of NO_x. All of the forgoing estimates include the refinery costs to produce the additional unleaded fuel that would be required, but not the costs to remove MMT. If the costs to remove MMT are included, it will add approximately $137/tonne for HC, $8/tonne for CO, and $113/tonne for NO_x.

Explicit cost-effectiveness comparisons between the cost per tonne of reductions of HC, CO, and NOx emissions from light-duty vehicles and the cost/tonne to achieve the same reductions from other sources could not be made due to the limitations of the available data. The only Canadian data available were for NOx reductions at thermal generating stations. Depending upon capacity utilization factors, expected station life, and the percent decrease in NOx emissions Ontario Hydro estimates that the cost per tonne of NOx removed ranges from $60 to $3 000 in 1984 dollars. For stations in Nova Scotia, New

Brunswick, Ontario, Saskatchewan, and Alberta, Environment Canada estimated that the cost per tonne of NO_x removed generally falls in the $115 to $360 range in 1982 dollars.

Estimates from United States studies of the cost per tonne to reduce HC, CO, and NOx emissions from other than light-duty vehicles vary widely depending upon the source and control strategy being considered. The cost per tonne of reducing emissions from light-duty vehicles by implementing the proposed standards are well within the range of the cost per tonne estimates to achieve emissions reductions from other sources.

Nonallocative Impacts

The nonallocative impacts considered in the analysis are: distribution effects, effect on dynamic efficiency, effect on market structure, effect on international trade, employment and output effects, and effect on inflation. Each of the nonallocative effects is summarized in turn below.

The income distribution effects of the proposed emission standards are: a higher level of absolute costs will be borne by the upper income groups, although lower income groups will pay a higher percentage of their income for the required emission controls.

In terms of regional distribution of costs and benefits, the costs of the required emission control systems will be borne by purchasers of cars across the country. The benefits are likely to be concentrated primarily in and downwind of larger cities, where there is a greater air pollution burden and automobile emissions are a significant factor in determining air quality.

Since the technology to meet the proposed standards is currently available for virtually all light-duty vehicles sold in Canada and is used for all vehicles sold in the United States, there is no requirement to develop new technology in Canada. As a result, the impact of the proposed regulations on dynamic

efficiency is minimal.

Currently, cars produced in Canadian and U.S. factories are equipped with at least two different types of emission control systems--one for Canada and one for the United States--on the same assembly lines. The proposed Canadian standards would eliminate the production of vehicles with emission systems designed specifically for the Canadian market. This may lead to increased production efficiencies and reduced inventory requirements.

There are two issues related to the effect on market structure. The first is that there are a number of off-shore manufacturers that sell cars in Canada, but not in the United States. Some of these importers do not have ready access to emission control systems that meet the proposed standards. Their cost of compliance will be higher than the competition. Since most of these importers compete mainly on price, any loss of price advantage will have an impact on sales.

The other issue is the effect of the proposed emission standards on the relative competitive position of the importers and domestic manufacturers. Based upon cost data that were submitted, offshore manufacturers are expecting to incur a higher cost of compliance than their North American counterparts. This apparent cost penalty will weaken the competitive position of the offshore manufacturers through either lost sales if retail prices are increased to cover the additional costs or lower profits if the additional costs are absorbed. The relatively higher increase in costs to the importers would tend to favour North American manufacturers.

It is estimated that 95 percent of the emission control equipment required to meet the proposed standards will be sourced outside of Canada. The effect on international trade attributable to these imports will be to increase Canada's imports of automotive products by up to $143 million (1983 dollars) per year or by just under 1 percent of the value of 1983 imports of automotive products.

The other issue related to international trade is the impact that implementing the proposed emission standards would have on the number of U.S. citizens that purchase cars in Canada. Under the terms of the Canada-U.S. Automotive Products Trade Agreement, U.S. citizens can purchase a car in Canada and import it to the United States, duty free, as long as it meets U.S. emission and safety standards. At the moment only about 15 percent of the cars sold in Canada meet U.S. emission standards and are eligible for import into the United States. If the proposed emission standards are implemented virtually all cars sold in Canada will be eligible for import to the United States.

Cars sold in Canada are currently less expensive than comparable models sold in the United States when differences in the Federal sales tax and exchange rate are taken into account. Prices for models with manufacturers' suggested retail prices less than $15 000 are generally 9 to 10 percent lower, on average, in Canada than in the United States. Cars currently sold in Canada that are eligible for import into the United States sell for about 5 percent less in Canada than the United States.

The price advantage and the availability of a wide choice of models should Canada's emission regulations be made substantially identical to those of the United States can reasonably be expected to result in an increase in the number of U.S. citizens buying cars in Canada. This will have a positive impact on Canada's balance of trade. However, in the absence of other measures to prohibit private importation of cars by U.S. citizens, the domestic motor vehicle manufacturers can be expected to come under intense pressure from U.S. dealers to increase car prices in Canada to levels that would discourage U.S. citizens from purchasing cars in Canada.

The outflow of funds of up to $143 million (1983 dollars) per year for additional emission control hardware that cannot be sourced in Canada will be equal to the sum of reduced expenditures on other items and reduced savings by car buyers. Estimates of the magnitude of these effects have been made with the use of Statistics Canada's Interprovincial Input-Output Model. The re-

sults of the input-output calculations suggest that the proposed emission standards will lead to a reduction in Canadian labour income of about $80 million/year, a reduction in Canadian GNP of about $120 million/year and possibly as many as 3 200 fewer jobs created in Canada each year than would be the case if the money was spent in Canada. It is emphasized that due to certain inadequacies in the input-output model and the changing structure of the Canadian economy since the model was constructed, these estimates of the impacts of increased imports are likely to be too high, possibly by a considerable amount.

Using the Statistics Canada definition of inflation, there will not be any impact on inflation if the proposed emission standards are implemented.

1. INTRODUCTION

Canada's current light-duty vehicle emission standards of 1.24 hydrocarbon (HC), 15.5 carbon monoxide (CO), and 1.92 nitrogen oxides (NOx) grams/kilometre (2.0 HC, 25 CO, 3.1 NOx grams/mile) came into effect in 1975. On October 14, 1978, notice was given in The Canada Gazette Part 1 that no amendments would be made to the emission standards prior to January 1, 1986, with the exception of the NOx standard of which an amendment might have been sought on or after January 1, 1984, if engineering and air quality data indicated a need for change. This was followed by an announcement in the February 20, 1982, issue of The Canada Gazette Part 1 that a Social and Economic Impact Assessment (SEIA) on the promulgation of a NOx emission standard of 0.62 g/km (1.0 g/mile) would be prepared.

The February 20, 1982, announcement was superseded by a notice published in The Canada Gazette Part 1 on September 18, 1982, in which notice was given that a SEIA on the promulgation of emission standards of 0.25 HC, 4.35 CO, and 0.62 NOx g/km (0.42 HC, 7.0 CO, 1.0 NOx g/mile) would be prepared.

The Canadian standards under consideration are substantially identical to the emission standards that have been in effect in 49 of the U. S. states since the beginning of the 1981 model year. (California is the exception; California's emissions standards are more stringent.) The 49-state emissions standards are referred to throughout this report as the "U. S. federal standards." The proposed CO standard of 4.35 g/km (7 g/mile) is higher than the U. S. federal CO standard of 2.1 g/km (3.4 g/miles). However, until 1983 the 3.4 g/mile standard was waived for some engine families in the United States, and the CO limit was set at 7.0 g/mile.

The terms of reference for this project specified that an analysis of the proposed emission standards be carried out in accordance with the methodologies outlined in Chapter 490, Appendix E, of Treasury Board Canada's Administrative Policy Manual. Chapter 490 specifies that, whenever

possible, benefit-cost analysis should be used to analyze the allocative effects (i.e., effect on market efficiency) of a proposed regulation. If benefit-cost analysis cannot be used, the cost-effectiveness methodology should be used.

An analysis of the impact of the proposed regulations on the following nonallocative factors, wherever appropriate, is required by Treasury Board: income distribution and regional balance, technological progress, market structure and competition, output and employment, balance-of-payments and international competitiveness, energy consumption, and inflation.

The effects of light-duty vehicle emissions on health, terrestrial ecosystems, aquatic ecosystems, man-made materials, and visibility are addressed in this report. The bulk of the discussion of the expected benefits concerns itself with benefits to human health. The major expected concrete benefits are in the area of human health. The legal authority for regulating motor vehicle emissions is based on protecting persons against injury, impairment of health, or death.

With the exception of the analysis of nonallocative effects, the material is drawn largely from four reports that were prepared as part of the background for this analysis. The topics covered by the four reports are: the effects of light-duty vehicle emissions, an assessment of air quality in four Canadian cities, the technology and costs to control light-duty vehicle emissions, and a review of methodologies to evaluate the benefits of reducing light-duty vehicle emissions. The four studies are briefly described in Chapter 2.

Additional information was provided by Environment Canada on a number of topics, including the contribution of light-duty vehicle emissions to ozone formation, the effects of nitrate on acidification of the aquatic system, and projections of light-duty vehicle emission factors and inventories.

Since the terms of reference for this study did not provide for any new work, the analysis is based on data from the sources listed above and the open literature. It was not possible to estimate dollar values for the benefits associated with light duty vehicle emission reductions based on currently available data. It was possible to identify adverse effects related to motor vehicle emissions and, in some cases, to estimate per-incident costs (e.g., the costs to treat one case of asthma). However, the degree to which the incidence of adverse effects would be reduced if the proposed light-duty vehicle emission standards are implemented cannot be determined at this time. A considerable amount of additional work on dose-response functions and the relationship between light-duty vehicle emissions and ambient concentrations of pollutants and deposition rates of nitrates is required before the information gaps will be filled. Even if sufficient resources were available, this work would take several years to complete.

The discussion on benefits, contained in Chapter 4, section 2, is qualitative due to the data deficiencies identified above.

The work on nonallocative effects was carried out by the authors, and is reported in Chapter 4, section 6.

2. BACKGROUND

2.1 Terms and Legal Authority

Levels of motor vehicle emissions are regulated under the authority of the Motor Vehicle Safety Act. Sections 4 and 7 of the act empower the Governor-in-Council (Cabinet) to make regulations affecting domestic manufacturers and importers of motor vehicles. Specifically, they "may prescribe safety standards for motor vehicles". The legal definition of "safety standard" is[1]:

> "standards regulating the design, identification, construction or functioning of motor vehicles and their components for the purpose of protecting persons against personal injury, impairment of health or death."

Under this authority, limits on motor vehicle emissions are prescribed as Canada Motor Vehicle Safety Standards (CMVSS). Permissible levels of exhaust emission and evaporative emissions are specified in CMVSS 1103 and CMVSS 1105. Motor vehicle particulate emissions are currently not regulated in Canada.

2.2 Related Studies

Four studies were commissioned by Environment Canada following publication of the notice of intent to prepare a SEIA of the proposed emission standards for light-duty vehicles.

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1. "An Act to Amend the Motor Vehicle Safety Act," assented to June 16, 1977, Vol. 2, No. 5, Canada Gazette Part III

A brief description of each of the studies follows.

1. Kolomeychuk et al, Effects of Automotive Emissions. Prime contractor: The Environmental Applications Group Limited. Final report; October 1983.

   This study documents the effects of carbon monoxide, nitrogen oxides, hydrocarbons, and diesel-exhaust particulates on: human health, terrestrial ecosystems, aquatic ecosystems, man-made materials, and visibility. The contribution of light duty vehicle emissions to the formation of acid rain and photochemical oxidants and their environmental effects are also considered in the report. The relationship between the dose of each substance to which the various populations at risk are exposed and their response was examined. Whenever possible, (quantitative) dose-response functions are provided.

2. Davis et al, Proposed Automobile Emission Standards, An Air Quality Assessment. Contractor: Concord Scientific Corporation. Final report; March 1984.

   The work carried out for this study includes a summary of air quality data collected between 1970 and 1982 in Toronto, Montreal, Edmonton, and Vancouver. The monthly and annual mean concentrations, percentile distributions of concentrations, and exceedances of national air quality objectives for each of carbon monoxide, nitrogen dioxide, ozone, total hydrocarbons, and total suspended particles are summarized. The report also includes an analysis of the contribution of automotive sources to urban emissions and a qualitative discussion of emissions in future years and their impact on air quality.

3. DeKany and Pilorusso, The Technology and Costs to Control Automotive Emissions in Canada. Prime contractor: Pilorusso Research Associates Inc. Final report; October 1983.

The main purposes of this study were to examine the technology required for more stringent control of automobile emissions and to determine the costs of the additional hardware that would be required to meet more stringent emission standards. Passenger cars and light trucks with gasoline and diesel engines were considered. The effects on emissions of cold-weather conditions and fuel consumption of more stringent emission controls are also discussed in the report.

4. Pilorusso et al, Review of the Methodologies To Evaluate the Benefits of Lowering the Levels of Automotive Emissions. Prime contractor: Pilorusso Research Associates Inc. Final report; April 1984.

   The benefits considered in this study are the expected reductions in the adverse effects of hydrocarbons (HC), carbon monoxide (CO), nitrogen oxides (NOx), and secondary substances, such as ozone and nitrates, if more stringent automotive emissions standards are implemented. The adverse effects are those identified in the first study described above. Human populations at risk nationally and in the four cities for which air quality projections were expected to be available--Toronto, Montreal, Edmonton, and Vancouver--are identified, and the size of each is estimated. The significance of personal activity patterns on exposure rates is examined. To the extent possible the value, in dollars, of the various effects is documented.

Several sections of this report are drawn largely from one or more of the four studies described above. Where this occurs, the study is identified. The references cited for specific findings or conclusions contained in the four studies are also cited in this report, although the original references were not always examined independently.

In addition to the contracted studies, four in-house studies were completed by Environment Canada staff. The titles of the studies are:

1. Elder, F. C., "Effects of Nitrate on the Acidification of the Aquatic System", Canada Centre for Inland Waters, Environment Canada, 1983.

2. Shaw, R. W., "The Atmospheric Pathway for Oxides of Nitrogen", Environmental Protection Service, Environment Canada, 1984.

3. Environmental Protection Service, "Light Duty Vehicle Emissions and the Oxidants Issue in Canada", Environment Canada report EPS 2/TS/3, May, 1984.

4. Environmental Protection Service, "Air Pollution Emissions and Control: Light Duty Vehicles", Environment Canada, 1984.

2.3 General Description of Air Pollution[2]

Air pollution may be considered as the presence in the atmosphere of a substance (gaseous or particulate), or mixture of substances, in sufficient amounts to produce a measurable effect on man, animals, vegetation, or materials. What is considered as a pollutant depends upon recognition of the substance (or substances) which cause adverse effects. Thus, air quality is poor if the concentration of a pollutant (or pollutants) in air exceeds the level at which it is considered to have an adverse effect.

Air pollutants can result from both natural and man-made sources. For example, hydrocarbons are emitted from vegetation. Oxides of nitrogen are emitted from soil and are produced by lightning. Carbon monoxide is formed from oxidation of methane which is produced by natural sources. Air pollution control strategies must take into account the relative contribution of natural and man-made emissions; but, in practical terms, emission reductions can be achieved only by controlling man-made sources.

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2. This section has been reproduced, with minor changes, from Kolomeychuk et al (1983).

There are two basic categories of air pollutants: primary and secondary. Primary pollutants, such as nitrogen oxides, carbon monoxide, and hydrocarbons, are released directly into the air from fuel combustion in stationary and mobile sources or from other processes. However, under the proper conditions, some of these primary pollutants can undergo chemical reactions and transformations in the atmosphere to produce secondary pollutants, such as ozone and nitric acid. The atmosphere serves as a delivery system from emission sources to the receptor. Once the substance is in the atmosphere, what happens to it depends on its physical and chemical characteristics rather than its origin (natural or man-made). Human activities that lead to air pollution are usually a result of overloading the atmosphere's capability to dilute, to transform emissions to innocuous compounds, or to recycle substances. The associated elevation of pollutant levels above that in clean air may cause adverse effects.

The final portion of the atmospheric cycle of a substance consists of several removal processes. These various processes serve to cleanse the atmosphere of accumulated material and deliver it to the surface of the earth. Removal processes may be classified into three types: wet removal, dry deposition, and chemical transformations. Wet removal is associated only with precipitation, whereas dry removal processes go on all the time. Chemical transformations in the atmosphere are thought of as a removal process since they transform one species into another.

One of the more important questions to be answered when a pollutant has been released into the atmosphere is what area will be affected. To attempt an answer, consideration must be given to the strength of the source, the way in which an effect is made manifest, whether the effects depend on airborne concentration or deposition, whether atmospheric processes are involved and meteorological considerations. Air pollution is, in the first instance, a local problem; however, the pollutants gradually mix into larger and larger volumes of air until they are distributed over very large areas. This leads to consideration of the properties of air pollutants on progressively larger spatial scales.

Air pollution problems may be grouped into four spatial scales: local (0 to 50 km), regional (50 to 500 km), continental (500 to 5000 km), and global (greater than 5000 km). One of the aspects that determines the zone of influence of an air pollutant is atmospheric residence time. An obvious requirement for a large-scale pollution problem is that a substance must have an atmospheric lifetime sufficiently long to be transported over long distances. A compound with a residence time of years, such as carbon dioxide, has time to mix throughout the atmosphere and become globally distributed. For compounds with shorter residence times, their potential zones of influence are reduced accordingly. For example, ozone can be expected to have a potential effect up to the continental scale, while dust raised from streets is only noticed up to several tens of metres. At continental scales, especially at mid-latitudes of the Northern Hemisphere, removal of some substances by precipitation becomes important.

On a local scale, pollutant levels can readily be distinguished from global, continental, and regional levels. For example, congestion of automobiles at intersections in downtown cores gives rise to levels of pollutants that are easily distinguishable from average levels over the city and the rural countryside.

Associated with the four scales of air pollution is the time for effects to be noticed at each level. On the local scale, deleterious effects could arise within hours while, at the opposite end of the spectrum, global effects may not be noticed even after decades.

2.4 Automotive Air Pollutants[3]

The exhaust of gasoline-powered vehicles is a source of three primary air pollutants: carbon monoxide (CO), oxides of nitrogen (NOx), and hydrocarbons (HC). Hydrocarbons are also released to the atmosphere through evaporation from the fuel tank and carburetor and during refueling. These substances

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3 This section has been summarized mainly from material in Kolomeychuk et al (1983).

contribute to two secondary air pollutants--ozone and acid precipitation. Ozone and other photochemical oxidants are produced by the action of sunlight on mixtures of NOx and HC; acid precipitation is usually attributed, in part, to NOx emissions.

Exhaust emissions from diesel-powered vehicles are similar to those from their gasoline-powered counterparts with a major exception. Diesel engines release substantial amounts of particles (soot) which contain organic compounds.

2.4.1 Carbon Monoxide

Carbon monoxide (CO) is a colourless, odourless, and tasteless gas. After carbon dioxide (CO_2), CO is the pollutant which is emitted into the air in the largest quantities. Man-made emissions are not the only sources; oxidation of methane in the atmosphere and oceans, and degradation of chlorophyll are natural sources. Overall, the man-made source of CO may be compared in magnitude with natural sources (Freyer 1979). The mean lifetime of carbon monoxide in the atmosphere is around 60 days, and its concentration in unpolluted air of the mid-latitudes of the Northern Hemisphere is around 0.23 to 0.29 mg/m^3 (0.20 to 0.25 ppm) (Enhalt 1981).

Concentrations of CO near man-made sources vary with time, season, and geographic locations. The dispersion and dilution of CO until it reaches background levels is affected by wind, atmospheric stability, vertical mixing height, and ambient temperature. The following automobile-related scenarios may result in unusually high ambient levels of CO (EPA 1979): (1) on a city freeway where traffic is stopped, CO levels may exceed 50 mg/m^3 (44 ppm); (2) inside a closed automobile where cigarettes are smoked, CO may exceed 100 mg/m^3 (87 ppm); and (3) in heavily-travelled vehicular tunnels, a one-hour maximum of 250 mg/m^3 (218 ppm) CO was recorded. The highest levels of carbon monoxide measured in Ontario in 1980 were at 381 Yonge Street, Toronto. This station, located close to a major traffic artery and

within an urban street canyon, measured an annual average of 4.8 mg/m^3 (4.2 ppm), a maximum 8-hour average of 29.9 mg/m^3 (26 ppm), and a maximum 1-hour average of 43.7 mg/m^3 (38 ppm) (MOE 1981). The annual arithmetic mean of monitors in rural locations was 0.12 mg/m^3 (0.1 ppm) which may be considered the background level.

2.4.2 Nitrogen Oxides

There are numerous oxides of nitrogen (NOx) but, for all practical purposes, NOx is taken to be the sum of nitric oxide (NO) and nitrogen dioxide (NO_2). Nitric oxide is a colourless, odourless gas which is only slightly soluble in water. In most urban atmospheres, NO is readily converted (oxidized) to NO_2. Nitrogen dioxide has a reddish-brown colour and a pungent odour. Several other compounds derived from NOx are of importance in air pollution. They include nitrates (NO_3^-), nitric acid (HNO_3), N-nitroso compounds, and organic compounds such as peroxyacetylnitrate (PAN).

Natural or clean-air levels of nitrogen oxides result from forest fires, lightning, microbial activity in soils and possibly oxidation of ammonia in the atmosphere. A unique difference between natural and man-made sources is that the former are widely distributed throughout the country while the latter are usually concentrated in urban and industrial centres. This concentration of anthropogenic sources leads to higher concentrations of NOx than are found in clean background air. The concentrations of NO and NO_2 in clean air over land are considered to be less than 0.001 ppm (Logan 1982).

Typical conditions for high NO_2 concentrations are usually a high flux of sunlight in combination with stagnant air. However, high concentrations of NO_2 occur in Northern Europe during relatively long periods of time characterized by an inversion, no wind, temperatures around or below 0°C, and relatively low photochemical activity. NO_2 concentrations of 188 to 564 µg/m^3 (0.1 to 0.3 ppm) in Gothenburg, Sweden, can last for several hours each day during an inversion period. The concentration of NO_2 in

polluted air in Canada, United States, and Europe sometimes exceeds 188 to 376 $\mu g/m^3$ (0.1 to 0.2 ppm) (Platt and Perner 1980, EPS 1981).

2.4.3 Hydrocarbons

The class of compounds known as hydrocarbons is unique among the emissions of automobiles. All pollutants except for hydrocarbons are believed to have direct effects on health, vegetation, or materials. Hydrocarbons mainly cause adverse effects indirectly through their contribution to the formation of photochemical oxidants. However, one member of this class, benzene (a suspected carcinogen), can be present in ambient air at levels that represent a potential adverse health effect (Tipton et al 1981).

2.4.4 Photochemical Oxidants

Ozone is the most abundant photochemical oxidant. Relatively high concentrations of ozone occur in the stratosphere where it is produced by ultraviolet radiation from the sun. It also occurs in high concentrations in polluted urban atmospheres and downwind of cities, being formed by the interaction of sunlight with ozone precursors: reactive hydrocarbons and nitrogen oxides.

Most of the reactions between precursors take place during the warmer months of the year when sunlight intensity is strongest. Conditions conducive to this type of pollution can occur up to a latitude of 60° North (i.e., close to the Arctic Circle) (Nieboer et al 1976). Hence, given the proper mixture and amount of precursors, there is the potential that ozone levels may be elevated at some time of the year throughout most of Canada. Because of the relatively long lifetime of ozone, it can be transported over long distances.

Relating changes in HC and NOx emissions to subsequent changes in ambient ozone concentrations involves considerable uncertainty. There is a general consensus that reducing hydrocarbon emissions would reduce the production of photochemical oxidants (OECD, 1979). An experimental model developed by

Glasson (1981) suggests that hydrocarbon reductions reduce O_3 in urban as well as downwind areas while NOx reduction alone increases O_3 in urban areas and has little effect on O_3 in downwind areas. This study pertains specificaly to Los Angeles. The results are not necessarily applicable to other urban centres.

There is substantial controversy on the role of NOx. Work carried out by others is consistent with Glasson's finding that a reduction in NOx emissions alone can result in an increase in ozone concentrations in urban areas (Derwent and Hov 1980, Innes 1981). However, in later work by Hov and Derwent (1981), it was found that the results depended upon the model that was used. One model predicted that NOx control or combined HC and NOx control is efficient with respect to O_3 reduction while another model predicted that HC control alone usually is more efficient than combined HC and NOx control. Furthermore, the latter model predicted that NOx control alone may increase the O_3 downwind burden.

The effect of NOx emission reductions in urban areas on O_3 concentrations in downwind, rural areas depends on whether the rural ozone concentration is primarily due to local production of ozone or to transport of ozone from the urban area. In most rural areas ozone production is limited by the amount of NOx available (Singh 1979). Therefore, any reduction in NOx transported to rural areas from urban areas would tend to decrease O_3 production. However, if local production of ozone in the rural area has little impact on ambient concentration, additional ozone produced in urban areas (as a result of reductions in NOx emissions) and transported to the rural area may result in increased O_3 concentrations in the rural area as well.

Agreement amongst the scientific community on an optimal control strategy for ozone is likely to remain elusive for at least a few more years. However, some general guidelines are evident from the work done to date. Hydrocarbon reductions alone appear to be the most efficient means of controlling O_3. Both HC and NOx reductions can be expected to reduce O_3 formation, but not as much as HC reductions alone. Reductions in NOx alone may result in increased O_3 concentrations in urban areas; the effect on the downwind ozone burden depends upon local conditions.

2.4.5 Diesel Exhaust Particulates

Comparisons of diesel-powered automobile emissions with its gasoline-powered counterpart generally show much higher concentrations of particulates, polycyclic aromatic hydrocarbon (PAH) compounds and sulphates, but lower emissions of carbon monoxide and total hydrocarbons. Sulphates and PAHs are associated with the particulate emissions. Light-duty gasoline-powered vehicles equipped with catalysts emit particulates at a rate of about 0.05 g/km, compared with 0.2-0.6 gm/km for diesels (Baires et al 1982). Approximately three-quarters of the particulate matter emitted from diesels is soot (National Research Council 1981).

2.4.6 Acidic Precipitation

There are many sources of the acids found in precipitation. Carbon dioxide contained in the atmosphere dissolves in rainwater to a certain extent and forms carbonic acid. At normal concentrations of CO_2 in the atmosphere, rainwater saturated with carbonic acid has a pH of about 5.6.[4] Other substances reaching the atmosphere shift the pH one way or another and determine the chemistry of natural precipitation. In areas where the soil is alkaline or basic, the incorporation of wind-eroded dust in precipitation may elevate pH levels to 6 or 7. Precipitation in coastal areas has enhanced concentrations of such sea-spray components as sodium and chloride. Gases such as sulphur dioxide, hydrogen sulphide, and nitrogen oxides which come from natural sources, can also alter the chemistry of precipitation. Sulphur dioxide and hydrogen sulphide are oxidized and hydrolyzed in the atmosphere to sulphuric acid; nitrogen oxides are similarly converted into nitric acid. If these acids are present in significant quantities, they can acidify precipitation.

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4. pH is a measure of acidity or alkalinity of a liquid; pH is presented on a logarithmic scale of 0 to 14, with 7 being a neutral state, 0 most acid, and 14 most alkaline.

Likens (1972) first indicated that nitric acid, resulting from the transformation of NOx, adds to the acidity of precipitation in the eastern United States. Brosset et al (1975) in Sweden showed that gaseous nitric acid increases acid deposition. Although sulphuric acid has been found to be the dominant source of acidity in Canada, the United States, and Europe, nitric acid accounts for almost one-third of all acid deposition, and there is evidence that this fraction is rising (Galloway and Likens 1981).

Galloway and Likens (1981) noted a significant seasonal pattern for the contribution of nitric acid to the acidity of precipitation. From measurements taken in the Hubbard Brook Experimental Forest they were able to determine that the maximum contribution of nitric acid was about 25 percent in the summer months and 45-55 percent in the winter months; for sulphuric acid, the summer maximum was 80-90 percent and the winter maximum, 70-80 percent.

Galloway and Dillon (1982), using continent-wide data, indicated that for the western half of North America there is enough nitrate (NO_3^-) to account for all the acidity of wet deposition. In a large region of eastern North America, NO_3^- can only account for less than 50 percent of the acidity. In the case of sulphuric acid (H_2SO_4), there is enough sulphate ($SO_4^=$) to account for all the acidity of wet deposition in Canada. While the absolute contribution of nitric acid (HNO_3) is uncertain, its levels appear to be increasing relative to sulphuric acid and in proportion to increased NOx emissions.

For mobile sources, consideration must be given to the probability that a significant fraction of NOx emissions are removed by dry deposition in the vicinity of sources (e.g., cities and highways). The ratio of wet-to-dry deposition increases with increasing distance from the source, and precipitation provides the major removal process in remote areas (Logan 1982).

There is one final aspect regarding the relative importance of nitric acid. Recently, fog has been investigated as a new aspect of the acid deposition phenomenon. Waldman et al (1982) reported pH values for fog water in southern California in the range 2.2-4.0, along with high nitrate-to-sulfate ratios. Brewer et al (1983) reported median pHs of 3.605 and 3.32 for 20 samples of mist and 10 samples of fog that were collected in the Los Angeles area. The acidity of the mist and fog is higher than the acidity of rain. Liljestrand and Morgan (1981) reported a mean pH of 4.49 for rain samples collected at a central Los Angeles site. The mist and fog samples had a ratio of $NO_3^-/SO_4^=$ equivalents of approximately 2 in contrast to Los Angeles rain data where the $NO_3^-/SO_4^=$ equivalents ratio is about 0.6 (Brewer et al 1983). Over 60 percent of the NOx emissions in the Los Angeles Basin are emitted at ground level from motor vehicles.

Wisniewski (1982) identified potential adverse effects of acid fog on vegetation and materials, but concluded that hard evidence on both the chemistry and effects of acidic events (dew, frost, and fog) was lacking.

2.5 Contribution of Light-Duty Vehicle Sources to Total Emissions[5]

The percentages of total emissions of HC, CO, and NOx that are from light duty vehicles are shown, for selected cities, in Table 2.1.

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5. The discussion in this section is based upon emission inventory data contained in Davis et al (1984).

Table 2.1

Light-Duty Vehicle Emissions 1978
('000 tonnes per year)

City	HC	CO	NOx
Toronto	70	740	45
% of total	40	68	27
Montreal	62	640	35
% of total	27	67	25
Vancouver	36	383	23
% of total	38	58	34
Edmonton	19	201	12
% of total	27	48	18

Light-duty vehicle sources account for a larger portion of total HC emissions in Toronto and Vancouver than in Montreal and Edmonton. Petroleum refining is a large source of HC emissions in Montreal and, as a result, light duty vehicle sources were proportionately lower than those in Toronto and Vancouver.

The majority of CO emissions in Toronto, Montreal, and Vancouver is from light-duty vehicles. Other major sources of CO are petroleum refining and nonhighway use of gasoline. Emissions of CO from nonhighway use of gasoline are proportionately higher in Edmonton than in the other cities, accounting for the relatively low percentage of CO emissions from light-duty vehicles.

Edmonton also shows a different pattern for NOx emissions, where only 18 percent of the total are from light-duty vehicles compared to 25 to 34 percent for the other cities. The apparent reason is that Edmonton has a proportionately higher volume of other mobile sources of NOx emissions than do the other three cities.

In all four cities, the largest apparent single source of HC, CO, and NOx emissions is passenger cars.

3. ADVERSE EFFECTS AND CONTROL OF AUTOMOTIVE EMISSIONS

3.1 National Ambient Air Quality Objectives[1]

The federal Clean Air Act of 1971 provided for the formulation of national ambient air quality objectives. These objectives are the goals toward which air pollution control programs are oriented. Canada's National Ambient Air Quality Objectives (NAAQO) do not have legal force unless they are promulgated as standards by provincial governments. These objectives are designed to protect public health and welfare, hence the purpose of revising automotive standards would be to ensure that objectives are not exceeded. Three levels of ambient air quality objectives are specified based on the recognition that there are a number of limits for each pollutant or combination of pollutants that could be established. These are:

Maximum Desirable Level
- ultimate goal for air quality.
- a basis for antidegradation policy for the unpolluted parts of the country and for the continuing development of control technology.

Maximum Acceptable Level
- provides adequate protection against effects on soil, water, vegetation, materials, visibility, personal comfort and well-being.
- realistic objective today for all parts of Canada.

  When this level is exceeded, control action by a regulatory agency is indicated.

Maximum Tolerable Level
- denotes a time-based concentration of an air contaminant beyond which, due to a diminishing margin of safety, appropriate action is required to protect the health of the general population.

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1. Sections 3.1 and 3.2 draw exclusively from Kolomeychuk et al (1983).

The concentrations of carbon monoxide (CO), ozone (O_3), and nitrogen dioxide (NO_2) for the three levels of the NAAQO are shown in Table 3.1.

Table 3.1

National Ambient Air Quality Objectives

Air Contaminant	Desirable Range	Acceptable Range	Tolerable Range
Carbon Monoxide (mg/m^3)			
1 hour average	0-15	15-35	-
8 hour average	0-6	6-15	15-20
24 hour average	-	-	-
Oxidants (ozone) ($\mu g/m^3$)			
1 hour average	0-100	100-160	160-300
24 hour average	0-30	30-50	-
annual arithmetic mean	-	0-30	-
Nitrogen Dioxide ($\mu g/m^3$)			
1 hour average	-	0-400	400-1000
24 hour average	-	0-200	-
annual arithmetic mean	0-60	60-100	-

Canada's 1-hour average maximum acceptable air quality objective of 35 mg/m^3 (31 ppm) for CO is higher than the World Health Organization's (WHO) recommended guideline and the United States proposed primary standards, both of which are 29 mg/m^3 (25 ppm), 1-hour average. United States primary air standards are set at a level that provides protection of public health.

The upper limits of the acceptable range for NOx are based on the following criteria:

1-hour
400 $\mu g/m^3$ (0.21 ppm)

a) the concentration is slightly below odour perception by the majority of young, healthy people at 410 $\mu g/m^3$ (0.22 ppm).

b) provides a safety margin of 10 for protection against air flow resistance in the presence of equal concentrations of SO_2.

c) lowest concentration shown to be harmful to animals is 433 $\mu g/m^3$ (0.23 ppm).

24-hour
200 $\mu g/m^3$ (.10 ppm) - no evidence of health effects at this level even when associated with a concentration of 260 $\mu g/m^3$ (0.10 ppm) SO_2.

1 year
100 $\mu g/m^3$ - based on the interpretation of the Chattanooga investigations prevalent at the time, i.e., slightly below the levels shown to have adverse effects; these studies suggested respiratory effects in children.

The criteria for setting desirable ozone objectives included the evaluation that no known acute or subacute health effects occur at the recommended levels.

The upper limit of the acceptable range of 160 $\mu g/m^3$ (80 ppb) for a 1-hour average exposure to ozone was established on the basis that this concentration was above the minimum level for impairment of performance of student athletes at 140 $\mu g/m^3$ (70 ppb) and experimental evidence indicating increasing susceptibility of mice to bacterial infection at 160 $\mu g/m^3$ (80 ppb) ozone for 30 hours. Canada's 1-hour maximum acceptable ozone limit is lower than the United States standard of 240 $\mu g/m^3$ (120 ppb), but higher than the WHO guideline of 120 $\mu g/m^3$ (60 ppb).

The threshold concentration at which adverse health effects are observed is not well known and to deal with this and other uncertainties, air quality objectives, guidelines, and standards are set to provide a margin of safety. The margin of safety provides for protection against health effects not yet identified or those identified, but not well understood.

3.2 Adverse Effects of Automotive Emissions

The adverse effects and the extent to which a cause-and-effect relationship between peak ambient concentrations of the substance and the adverse effect has been established are summarized in Table 3.2. Although concentrations at which there is an increased probability of occurrence of each effect are noted in Table 3.2, it is stressed that the numbers do not represent absolute thresholds. In most cases thresholds are not well established and some of the effects may occur at lower concentrations than those noted.

Table 3.2

Summary of Possible Major Effects Of Automotive Emissions

Substance/Effect	Cause-and-Effect Relationships Health Effects
Carbon Monoxide	
1. Myocardial infarction (heart attack)	Increased probability of occurrence when carboxyhaemoglobin (COHb) reaches 3%; minimum ambient concentration of CO: 22.9 mg/m^3 (20 ppm), 8 hours' exposure.
2. Aggravation of angina pectoris (attacks of sharp chest pain due to inadequate blood supply)	Increased probability of occurrence: COHb of 2.5-3%; minimum ambient concentration of CO of 19.5 mg/m^3 (17 ppm), 8 hours' exposure.

Table 3.2, continued

Summary of Possible Major Effects Of Automotive Emissions

Substance/Effect	Cause-and-Effect Relationships
Health Effects	
Carbon Monoxide	
3. Intermittent claudication (leg pain or limping due to inadequate blood supply	Same as #2, above.
4. Decreased work capacity (breathlessness, exhaustion)	Increased probability of occurrence: COHb of >4%; minimum ambient concentration of CO of 32.1 mg/m^3 (28 ppm) 8 hours' exposure.
5. Impaired vigilance (visual, auditory, reaction time, time estimation, etc.)	Increased probability of occurrence: COHb of 3-4%; minimum ambient concentration of CO of 22.9 mg/m^3 (20 ppm), 8 hours' exposure.
6. Effect on fetal development (decreased birth weight, increased perinatal deaths, and decreased weight gain in newborns; reduced learning and general abilities in later childhood)	Increased probability of occurrence: maternal COHb >4%, fetal COHb of 7%; minimum ambient concentration of CO of 32.1 mg/m^3 (28 ppm), 8 hours' exposure.

Table 3.2, continued

Summary of Possible Major Effects Of Automotive Emissions

Substance/Effect	Cause-and-Effect Relationships
Health Effects	
Nitrogen Dioxide	
1. Increased incidence of respiratory symptoms (chest discomfort, cough, headache, sore throat, eye irritation, shortness of breath)	Increased probability of occurrence for young children at ambient concentration of NO_2 of 752-1880 $\mu g/m^3$ (0.4-1 ppm); healthy adults at NO_2 of 4700 $\mu g/m^3$ (2.5 ppm).
2. Aggravation of respiratory disease (attacks of asthma, bronchitis or emphysema)	Increased probability of occurrence at ambient concentration of NO_2 of 940 $\mu g/m^3$ (0.5 ppm), although threshold level is not well established.
Ozone	
1. Increased incidence of respiratory symptoms (same as NO_2)	Increased probability of occurrence at ambient concentration of ozone (O_3) of 300-500 $\mu g/m^3$ (0.15-0.25 ppm), or lower.
2. Aggravation of respiratory disease (same as NO_2)	Increased probability of occurrence at ambient concentration of O_3 of 300 $\mu g/m^3$ (0.15 ppm), or lower.
3. Decreased resistance to infection	Increased probability of occurrence at ambient concentration of O_3 of 360 $\mu g/m^3$ (0.18 ppm).

Table 3.2, continued

Summary of Possible Major Effects Of Automotive Emissions

Substance/Effect	Cause-and-Effect Relationships
Health Effects	
Ozone	
4. Decreased performance	Increased probability of occurrence at ambient concentration of O_3 of 200-300 $\mu g/m^3$ (0.10-0.15 ppm).
Terrestrial Ecosystems	
Ozone	
1. Reduced production of forests	Significant effect has been observed for sensitive species with ambient concentrations of O_3 varying from 60-80 to 200-240 $\mu g/m^3$ (0.03-0.04 to 0.10-0.12 ppm) 24-hour average during growing season.
2. Changes in forest metabolism, growth, and composition	Effects are observed in areas where 1-hour maximum ambient concentration of O_3 reaches 100-800 $\mu g/m^3$ (0.05 to 0.40 ppm).
3. Reduced yield and value of agricultural crops	Significant effects are observed for sensitive plant species at 7-hour midday seasonal average ambient concentration of O_3 of 84-200 $\mu g/m^3$ (0.042-0.100 ppm).

Table 3.2, continued

Summary of Possible Major Effects Of Automotive Emissions

Substance/Effect	Cause-and-Effect Relationships
Aquatic Ecosystems	
Oxides of Nitrogen	
1. Contribution of nitrate (NO_3^-) and nitric acid (HNO_3) to acidic precipitation	Cause-and-effect relationship not well understood. HNO_3 contributes to freshwater acidification during spring melt of snowpack containing nitrate.
Man-Made Materials	
Nitrogen Dioxide	
1. Fading of fabric dyes	Significant fading of some fabrics is observed on 12 weeks' exposure at a concentration of 94 $\mu g/m^3$ (0.05 ppm) NO_2 with high humidity (90%) and high temperature (30°C).
2. Yellowing of some white fabrics	Effect documented in chamber studies at a concentration of NO_2 of 37.6 $\mu g/m^3$ (0.02 ppm) and exposure of 8 hours.
3. Cracking of springs made of nickel-brass alloy	NO_2 may lower the humidity threshold for corrosion.

Table 3.2, continued

Summary of Possible Major Effects Of Automotive Emissions

Substance/Effect	Cause-and-Effect Relationship
Man-Made Materials	
Ozone	
1. Cracking of tire sidewalls and loss of ply adhesion	Cracking would occur at ambient concentrations if antiozonants were not added to the rubber compound.
Visibility	
Oxides of Nitrogen	
1. NO_2 may cause a brownish haze in a polluted layer when viewed against a relatively clean sky	Color shift should be detectable at a concentration-visual range product of 564 $\mu g/m^3$-km (0.3 ppm-km) NO_2.
2. Nitrate fine particulates contribute to light extinction	Of little consequence generally but may be significant in certain areas.

Those possible adverse effects of automotive emissions that have been observed, or are suspected, are listed in Table 3.2. There may be other effects that have not yet been observed or for which the scientific evidence is too tenuous to even establish a strong suspicion of causality. Therefore, the effects listed in the table likely do not represent all of the possible adverse effects of automotive emissions.

Nitric acid may play a significant role in short-term acidification of streams and lakes. McLean (1981) reviewed North American and Scandinavian studies and found that nitric acid tends to accumulate in the snowpack over the winter. During rapid melt-runoff in the spring, higher than normal amounts of nitric acid are carried into lakes and streams at a time of particular sensitivity to the life cycles of many aquatic species. This phenomenon has become known as "spring shock". McLean's assessment was that nitric acid contributes slightly less than sulphuric acid to observed drops in stream and lake pH (i.e., increased acidity) during snowmelt.

Aquatic and terrestrial ecosystems may be expected to suffer adverse effects from acid rain. However, there is little available evidence to suggest that automotive emissions and their secondary compounds, formed by photochemical reactions, contribute significantly to the long-term adverse effects of acid rain. The available evidence would suggest that the levels of nitric acid (HNO_3) and nitrates (NO_3^-) in acid rain increase with increased NOx emissions (see section 2.4.6), but the environment is able to assimilate HNO_3 and NO_3^- during the growing season without causing apparent damage. Studies by Galloway and Dillon (1982) and McLean (1981) suggest that interaction of nitrate ions with soils and vegetation allows nitric acid to be largely assimilated by the terrestrial portion of the watershed (MOI 1983). Similarly, under most conditions nitrate falling directly on a lake would be easily assimilated by aquatic vegetation. Since there are few comparable terrestrial sinks for sulphur, sulphate ($SO_4^=$) is a conservative ion whose export-to-surface waters is directly related to deposition in precipitation. Therefore, most of the observed increases in acidity are due to changes in sulphate concentrations. There is no strong evidence at present for anticipating any appreciable reduction in long-term lake or stream acidification from a reduction in nitrate inputs.

After reviewing the evidence on the effects of acidic precipitation on terrestrial ecosystems, Kolomeychuk et al (1983) concluded: "No documented losses of agricultural crops, forests, or horticultural plants appear to have been clearly identified with acid precipitation in North America."

In a recent study, Dendron Resource Surveys Ltd. (1984) reported that prolonged exposure to acidic precipitation may result in tree mortality, loss of vigour and growth, and irreversible damage to soil and other components of the forest ecosystem. Although acidic precipitation has been implicated in widespread damage to forests in Germany, in Canada there is, at present, no evidence of significant forest damage through the long-range transport of air pollutants (Dendron 1984).

It must be emphasized that the findings stated above reflect a lack of concrete evidence rather than an evaluation of a large data base that would support an unequivocal conclusion that there does not exist a cause-and-effect relationship between automotive emissions and the long term, adverse effects caused by acidic precipitation. The relationship is not well understood in quantitative terms. Therefore, no attempt is made in this report to establish dollar values for damage that may be caused by the contribution of light-duty vehicle emissions to acidic precipitation.

3.3 Automotive Emissions Control Technology[2]

3.3.1 Current Gasoline-Engined Passenger Cars

Four basic types of emission control systems are used to meet the current emissions standards of 1.24 HC, 15.5 CO, 1.92 NOx grams per kilometre (2.0 HC, 25 CO, 3.1 NOx grams per mile).

Noncatalyst

The typical system in this category uses enleanment of air/fuel ratio, exhaust gas recirculation (EGR), and, usually, injection or aspiration of air into the exhaust manifold using either belt driven air pumps or pulse air (reed valve) devices. Conventional carburetors or mechanical fuel injection (low pressure, manifold) are generally used for fuel metering. Electronic (multi-point, manifold) fuel injection systems are occasionally used in luxury or high performance models. Compared to preemission control engines, engine modifications (such as reduced compression ratio, revised combustion chamber geometry, retarded spark timing, modified valve overlap, modified intake and exhaust manifolds, etc.) are common. Breakerless high energy ignition and heated intake manifolds (early fuel evaporation) are also commonly used.

Oxidation Catalyst

The typical system in this category uses the same components as in the noncatalyst system above, except, an oxidation catalyst is added. Oxidation catalyst systems used in Canada by the North American manufacturers are derived from systems used in the United States prior to 1981. These systems generally allow ample margin in meeting Canadian emission standards and provide better fuel economy than noncatalyst systems.

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2. This section draws largely from DeKany and Pilorusso (1983).

U.S. Federal System

The typical system in this category uses the emission control system employed by the manufacturer to meet the 1983 U.S. federal emission control standards of .25 HC, 2.1 CO, 0.62 NOx grams per kilometre (.41 HC, 3.4 CO, 1.0 NOx grams/mile). Usually this system uses a three-way catalyst or three-way plus oxidation catalyst in conjunction with electronically-controlled feedback fuel management. The latter consists of an electronic control module, oxygen sensor, and a means of providing variable air fuel ratio control, such as, variable jet carburetor or electronic fuel injection (throttle body or multi-point). In some cases, the electronic control system is expanded to provide electronic control of spark timing, EGR, and air injection. Air injection is also common on these systems. These systems offer substantially more emission control margin than necessary for the current Canadian standards.

Special

A few manufacturers offer models with unique approaches to the control of emissions, such as the stratified charge engine (CVCC) of Honda, the NAPS-Z engine of Nissan, and the MCA-Jet offered by Mitsubishi. In general, these systems offer greater emission control margin than necessary for the current Canadian standards.

Approximately 98.5 percent of passenger cars sold in Canada are equipped with gasoline engines. Passenger cars with diesel engines have accounted for 1 to 2 percent of total registrations in recent years. New car registration data compiled by R.L. Polk & Co. Ltd., show that approximately 1.5 percent of cars registered in 1983 have diesel engines. A small number of cars are equipped to run on propane.

Emission control systems on gasoline-engined passenger cars sold in Canada in 1983 were approximately as follows: 20% noncatalyst, 63% oxidation catalyst, 15% U.S. federal systems, and 2% special systems.

3.3.2 Current Gasoline-Engined Light Trucks

The general trend of emission control technology usage for automobiles also holds true for light trucks. Unique Canadian systems are used whenever production volumes are large enough to justify them. Otherwise the U.S. federal systems for light trucks are used. Two generic types of emission control systems are currently utilized: noncatalyst systems and oxidation catalyst systems. The hardware composition of these systems is as already described above for passenger cars. Since the U.S. emission standards for light trucks are less stringent than for passenger cars--U.S. light truck standards are: 1.05 HC, 11.16 CO, 1.43 NOx g/km (1.7 HC, 18 CO, 2.3 NOx g/mile)--the use of three-way catalysts is unnecessary. Light-truck oxidation catalyst systems used in Canada are generally of two types--U.S. light-truck federal system or a basic U.S. federal system--but with a lower volume of catalyst and lower noble metal loading.

3.3.3 Current Diesel-Engined Passenger Cars and Light Trucks

The diesel engine is an inherently low emitter of hydrocarbons and carbon monoxide. The current HC and CO standards are usually easily attained by diesel engines.

With minor design changes diesel engines have little difficulty in meeting NOx standards that are no more stringent than 1.24 g/km (2.0 g/mile). Therefore diesel-engined passenger cars and light trucks have no difficulty in meeting the current Canadian standard of 1.92 g/km (3.1 g/mile) NOx. Few, if any, diesel-engined passenger cars and light trucks sold in Canada require any additional hardware to meet current emission standards.

3.3.4 Emission Control Systems to Meet Proposed Standards

DeKany and Pilorusso (1983) examined the technology and costs to control emissions at three levels:

Level	Standard - Grams/Kilometre			Standard - Grams/Mile		
	HC	CO	NOx	HC	CO	NOx
Current	1.24	15.5	1.92	2.0	25	3.1
Intermediate	0.93	9.3	1.24	1.5	15	2.0
Proposed	0.25	4.35	0.62	0.41	7	1.0

The intermediate levels are equivalent to the 1977 U.S. emissions standards. The proposed levels are those published in the September 18, 1982, Canada Gazette, Part 1 notice.

Most current Canadian emission control systems are not capable of meeting the intermediate standards. If the intermediate standards were implemented, manufacturers would have the choice of either developing new Canadian systems or equipping cars for sale in Canada with U.S. federal systems. Most manufacturers stated that, if Canadian standards were set at any level more stringent than current levels, they would find it more cost-effective to install U.S. federal systems than to develop unique Canadian systems. This leaves, in effect, two options: the status quo and the proposed standards. The intermediate standards will not be considered further.

Gasoline-Engined Passenger Cars

It is possible, in some cases, to meet the proposed standards without using a closed loop, three-way catalyst system. Oxidation catalyst systems similar to those currently used, but with larger catalyst volumes and increased noble metal loadings, are capable of meeting the proposed standards for some lightweight, 4-cylinder-engine passenger car models. During the first year that the current U.S. federal standards came into effect in the United States, 1981, approximately 14 percent of the fleet was equipped with oxidation catalyst systems (Wagner et al 1982). However, with few exceptions, manufacturers indicated that the closed loop, three-way catalyst system would be required to meet the proposed Canadian standards.

Diesel-Engined Passenger Cars

Unlike gasoline-engined passenger cars, currently available emissions control systems for diesel-engined passenger cars cannot meet the proposed emission standards. The problem is meeting the 0.62 g/km NOx standard. Virtually all of the manufacturers that sell diesel-engined passenger cars in the United States have obtained waivers to allow NOx emission above the 1.0 g/mile (0.62 g/km) level specified by the United States Clean Air Act.

Light Trucks

The notice in the September 18, 1982, Canada Gazette, Part 1, announcing the proposed emission standards for light-duty vehicles did not differentiate between passenger cars and light trucks. If the proposed standards were applied without changes to light trucks, the impact would be considerable. The proposed standards are more stringent than those currently in effect for light trucks in 49 states of the United States, and

the only proven light-truck emission control systems that would meet the proposed standards would be recalibrated versions of those used in California.

If, on the other hand, Canadian light-truck standards were set at current U.S. levels of 1.05 HC, 11.16 CO, 1.43 NOx g/km (1.7 HC, 18 CO, 2.3 NOx g/mile), the changes required would be relatively minor. Many Canadian light-truck models already use the U.S. emission control system. No emission controls would be required in the case of diesel-engined light trucks because the uncontrolled diesel engine is capable of meeting the standard.

3.3.5 Fuel Consumption

The manufacturers uniformly estimate a fuel-consumption penalty for their passenger car fleets if the proposed standards are adopted. Generally, this estimate is based on comparisons of vehicles with unique Canadian emission control calibrations against similar vehicles employing U.S. federal emission control systems. The main technical explanation given by the manufacturers for the poorer performance of the U.S. federal systems, more specifically the three-way catalyst system, is that the air/fuel ratios of these systems[3] must be calibrated at or slightly fuel-rich of the stoichiometric point; whereas the optimum calibration for minimum brake specific fuel consumption is lean of the stoichiometric point. While this argument is technically correct, air/fuel ratio is only one of many emission control system and powertrain design and operating parameters that can effect fuel consumption.

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3. The stoichiometric point is where the ratio of fuel to air is the exact ratio to produce complete combustion without producing leftover fuel or air.

Manufacturers' estimates of the magnitude of the fuel-economy penalty generally fall in the 3 to 5 percent range. However, studies comparing the fuel consumption of cars equipped with U.S. federal systems and otherwise identical cars equipped with Canadian systems have found no statistically significant difference in fuel-consumption between the vehicles, especially when a broad temperature range was studied and when road fuel consumption measurement procedures were used (Polak et al 1984).

From a purely technical consideration, there should be no significant difference in fuel consumption between unique Canadian calibrations currently sold and those calibrations required for compliance with the proposed standards. The current Canadian emission control systems are largely open-loop, non-electronic systems, with many already employing catalytic control. The U.S. federal systems, which will be the precursors of the systems needed to meet the proposed standards, are closed-loop, electronically-controlled systems. Any fuel-consumption difference caused by the richer fuel/air ratio of the U.S. federal system can be offset by the benefits of electronic control and, if necessary, by downsizing engine displacement (operation at the richer fuel/air ratios of the federal system increases engine power). Further, greater application of fast burn techniques and expansion of electronic control to additional engine systems (e.g., electronic controls of air, spark, and EGR, and improved fuel-injection systems) will result in continued improvement in fuel consumption of federal systems.

However, the manufacturers are quite correct in emphasizing the necessity to make trade-offs among emissions, fuel consumption, cost, driveability, and other performance parameters. In turn, this trade-off is affected by conditions that are often outside the control of the manufacturers' technical staffs, such as the

health of the automotive market and the general economy, and the time and resources available to the manufacturer. At current emission control levels, most manufacturers possess, because of prior U.S. experience, a large base of certified designs from which to select and optimize fuel economy without much expenditure. At proposed levels, most, if not all, manufacturers will have to use the U.S. emission control systems in use at the time the proposed levels are adopted. Thus, the trade-offs will be dictated mainly by market and economic trends in the United States. If the U.S. buyer continues to place emphasis on fuel economy, then fuel consumption of Canadian vehicles meeting the proposed standards should be equal to or better than current Canadian offerings.

DeKany and Pilorusso (1983) concluded that the fuel consumption impact of implementing the proposed standards will be minimal and slightly positive--a fuel consumption improvement of 1 to 2 percent. This was based on the assumption that improvements in U.S. federal systems between 1983 and model year 1986 would bring about a 5 to 6 percent improvement in fuel economy. An implicit assumption underlying this conclusion was that, if the current Canadian emission standards are maintained, the fuel economy of Canadian cars would not improve over the next 2 to 3 years. However, subsequent to the time that DeKany and Pilorusso's report was issued, a number of manufacturers have pointed out that Canadian calibrations are based upon U.S. federal systems and many of the improvements in fuel economy made for U.S. federal systems would be carried through to Canadian systems. The manufacturers would also continue to calibrate Canadian systems for optimal fuel economy whenever volumes were large enough to justify development and certification costs. Thus, a 1 to 2 percent improvement in fuel economy seems unlikely if the proposed standards are implemented.

The available evidence would suggest that the overall fuel consumption impact of the proposed emission standards is small. For the purpose of estimating the cost of the fuel consumption impact over the years 1987-1996, it will be assumed to be zero for the portion of the light-duty vehicle fleet that is currently equipped with catalyst systems. For the portion of the light-duty vehicle fleet that currently does not have a catalyst system and operates on leaded fuel, switching to a catalyst system and unleaded fuel is expected to result in improved fuel economy. The magnitude of the fuel savings is discussed in section 4.3.

3.3.6 Durability, Maintenance, Driveability and Other Impacts

The manufacturers responding to a survey carried out by DeKany and Pilorusso (1983) uniformly reported that they do not expect any impact on durability, maintenance, or driveability as a result of the proposed standards. For example, General Motors responded, "We do not anticipate any problems obtaining satisfactory driveability and durability with any emission control systems we are considering for the proposed emission standards. We would compromise fuel efficiency to the extent necessary to meet the applicable emission standards and obtain satisfactory driveability."

The proposed standards will require increased use of unleaded fuels and stainless steel components so that exhaust system durabilty will be enhanced. Since these systems will be calibrated richer and employ advanced electronic controls, driveability should also be enhanced--including cold weather starting and performance.

The use of unleaded gasoline reduces engine maintenance requirements. There is less corrosion of the exhaust system resulting in fewer muffler and exhaust pipe replacements. There is less fouling and corrosion of spark plugs which result in better engine performance and longer spark plug life. Engine wear is reduced due to less corrosion which allows longer periods between oil changes.

On the other hand, the emission control systems required to meet the proposed standards are considerably more complex and use more advanced technology than current Canadian emission control systems. Subsequent to the survey carried out by DeKany and Pilorusso (1983) the Motor Vehicle Manufacturers' Association stated that maintenance problems and costs will increase for motor vehicles equipped to meet the proposed standards (MVMA 1984).

The only durability concern is the inadvertent or purposeful use of leaded fuels in vehicles that are designed to run on unleaded fuel. This type of misfuelling causes poisoning of the catalyst, clogging of EGR valves, and deterioration in the performance of the oxygen sensor. Use of leaded fuel can cause emission control system failure and emissions to increase significantly. Failure of the oxygen sensor may also result in increased fuel consumption.

4. ANALYSIS OF ALLOCATIVE AND NONALLOCATIVE EFFECTS

Allocative costs are defined as "opportunities for production or consumption forgone because of projects undertaken"; allocative benefits are "those favourable consequences of projects which represent opportunities to increase production or consumption" (Treasury Board 1976). According to Treasury Board (1976), "It is considered that a project is efficient and should be undertaken if its allocative benefits exceed its allocative costs."

Estimates of the allocative costs are presented and the allocative benefits are discussed in this chapter. It is possible to identify and calculate most of the costs, but a complete evaluation of benefits is not possible due to limitations in the data.

The nonallocative effects that are discussed in this chapter are: distribution effects, effect on dynamic efficiency, effect on market structure, effect on international trade, employment and output effects, and effect upon inflation.

Adoption of the proposed emission standards will require that all light-duty vehicles be equipped with catalyst sytems. As a result, the portion of new, gasoline-powered light-duty vehicles that require unleaded fuel will increase from about 80 percent to 100 percent. A summary of the allocative costs and benefits of the shift in demand from leaded to unleaded gasoline is presented in this chapter.

4.1 Costs of Emission Control Systems

All of the cost data presented in this section are taken from the report, The Technology and Costs to Control Automotive Emissions in Canada (DeKany and Pilorusso 1983). The costs of emission control systems presented in the report come from two sources. The first is work done on behalf of the United States Environmental Protection Agency (EPA) by L. H. Lindgren in 1977 (Lindgren 1978). Some of Lindgren's estimates were subsequently updated in 1980-81 by EPA staff (Hellman 1981.) At the same time new cost estimates were made by EPA staff for components for which the design was finalized between 1977 and 1980, e.g., electronic control units.

The other source of cost data was domestic motor vehicle manufacturers and importers. These data were provided as the increment in the retail price of a particular make and model of motor vehicle if the emission control system used in 1983 was replaced with a system that would meet the proposed emission standards.

The emission control systems required to meet the proposed Canadian emission standards are substantially identical to those used to meet the current emissions standards for 49 of the U. S. states. These systems were first used on production vehicles in 1981. Since the Lindgren and EPA estimates were prepared prior to 1981 and the cost data provided by manufacturers and importers are based on 1983 costs, neither set of data fully reflects decreases in costs that are likely to be made possible as the technology is refined and as manufacturers gain more experience with it. No attempt is made to account for "learning-curve" effects on the costs that are presented here. However, it can be reasonably expected that the real costs of emission control systems in the years 1987 to 1996 will be less than those shown in this section.

From the available cost and other data, the average retail price equivalent of the emission control equipment was estimated to be $235 per car. The average retail price equivalent can be disaggregated as shown in Table 4.1.

DeKany and Pilorusso (1983) showed disaggregated retail price equivalents for two markup factors, 2.7 and 1.93. (The markup factor is a composite of parts manufacturers', vehicle manufacturers', and dealers' markups.) The higher markup factor, 2.7, is based upon assumed markups at each level (Lindgren 1978.) The lower markup factor, 1.93, is based upon EPA's analysis of actual cost data, and probably more accurately reflects true costs and markups. The disaggregated retail price equivalent shown in Table 4.1 is based upon the lower markup factor.

Table 4.1

Components Of The Incremental Retail Price Equivalent (1983 $)

Component	Percent	Amount per Vehicle
Plant manufacturing cost		
o material	31.4	74
o labour	9.6	24
o plant overhead	3.8	9
Subtotal	44.8	107
Component vendor markup	10.3	24
Tooling expense	2.9	7
Assembly, engine and body modifications	2.0	5
Total--vehicle manufacturer cost	60.0	143*
Vehicle manufacturer markup	13.8	32
Federal sales tax	6.6	16
Dealer markup	13.8	32
Research and development	5.7	13
Total--Retail price equivalent	100.0%	$ 236**

* Does not equal 141, (60 percent of 235) due to rounding.

** Total does not equal $235 due to rounding.

The vehicle assembler and dealer markups applied to the cost of emission control systems are the "normal" industry markups. It is by no means certain that all of the automotive manufacturers will apply full markups to their incremental costs; some manufacturers may pass the incremental costs through to the consumer without markups, others may have to absorb the incremental costs in order to remain price-competitive in the market. However, since the proposed emission standards represent a permanent increase in costs to all of the motor vehicle manufacturers and importers serving the Canadian market, the incremental costs for the emission control hardware will become part of the cost base to which normal overheads are applied for accounting purposes.

The motor vehicle manufacturers have a number of options for dealing with their costs for emission control hardware: they can absorb the costs and accept lower gross profit margins, they can increase prices in other markets to compensate for lower margins in Canada, they can increase prices in Canada, or any combination of the foregoing. In the short term, individual manufacturers will set retail prices based upon the degree of price sensitivity in the marketplace, the amount of price competition, how well their products are selling relative to competitors, and other factors. Therefore in the short term, retail prices may bear little relationship to the manufacturers' cost of parts and materials. However, implementing emission control standards do incur real costs and, ultimately, the consumer can be expected to bear those costs.

For the reasons discussed above--the potential for decreases in the costs of emission control hardware and the possibility that full, normal markups will not be immediately applied by motor vehicle manufacturers to their incremental costs for the hardware--the retail price equivalent data that are shown in the following pages should be regarded as the <u>maximum</u> increases in motor vehicle prices to the consumer if the proposed regulations are implemented.

4.1.1 Passenger Cars

For the purpose of calculating costs, it is assumed that the proposed standards will come into effect on January 1, 1987. From registration statistics (Automotive News 1983) it can be inferred that the in-use fleet renews itself every 10 to 11 years. A small number of passenger cars more than 10 to 11 years old will remain in the active fleet, but the number of vehicle kilometres travelled (VKmT) by them will represent a very small portion of fleet VKmT. Therefore, for the purpose of calculating costs, a 10-year period will be used, starting in 1987 and ending in 1996.

Retail Sales

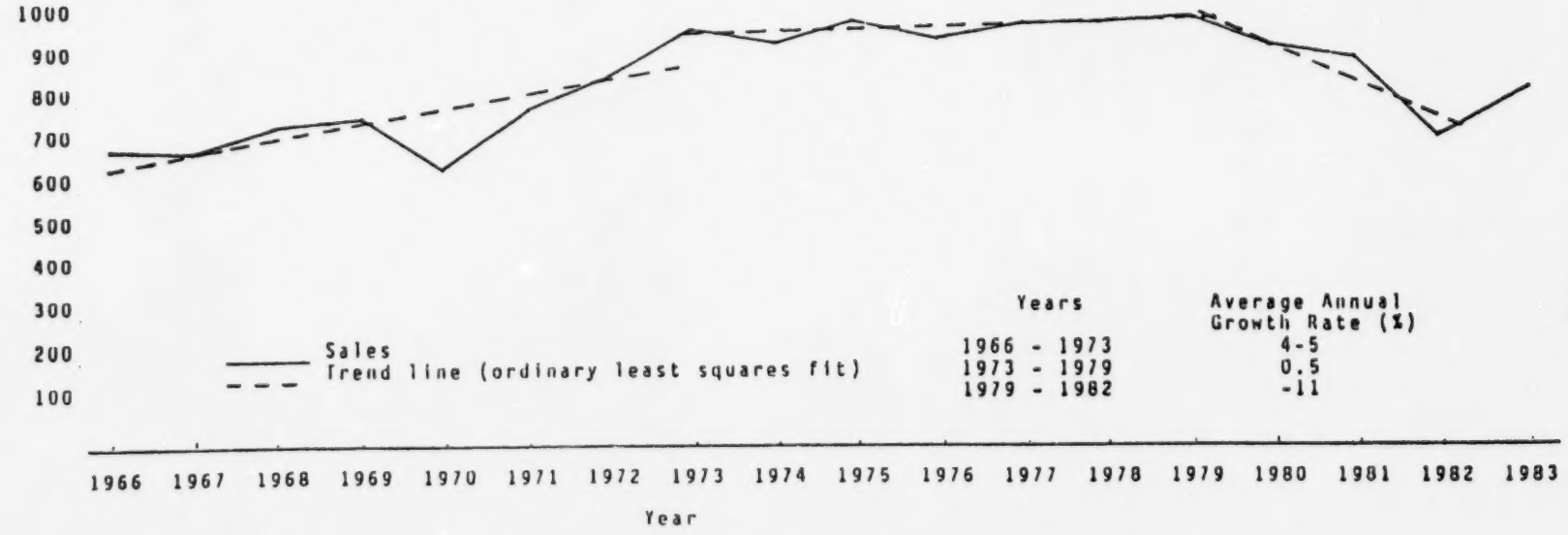
Retail sales of passenger cars from 1966 to 1983 are shown in Figure 4.1. The last time period during which there was significant growth in the market was 1966-73. Average annual growth was 4 to 5 percent during those years. The main factor underlying the market growth was the increase in the size of cohort of persons aged 18-24, which includes many first-time car buyers. That is, the baby-boom generation was coming of age. The market reached a plateau of almost a million passenger car sales per year in 1973 and remained remarkably constant until 1980. Sales between 1973 and 1979, inclusive, varied between about 943 000 and 1 003 000, with average annual growth of approximately 0.5 percent. From 1980 to 1982, sales fell at the rate of 11 percent per year for 3 consecutive years.

A precise forecast of passenger car sales during the years 1987 to 1996 is not necessary for the present purpose. The estimates of passenger car sales shown in Table 4.2 are based upon the following assumptions:

- During 1985 annual sales will have returned to the 1973 to 1979 average of about 975 000 passenger cars.

Fig. 4.1

Retail Sales of Passenger Cars



Years	Average Annual Growth Rate (%)
1966 - 1973	4-5
1973 - 1979	0.5
1979 - 1982	-11

Source: Statistics Canada, Motor Vehicle Sales, Cat. No. 63-007

- Compound annual growth from 1986 to 1992, inclusive, will average 0.5 percent. The number of first-time car buyers will be declining during that period, but replacement demand should remain high since most of the cars purchased between 1973 to 1979 will be retired from the fleet by the early 1990s.

- Sales will decline by 10 percent from year-earlier levels in each of 1993 and 1994, and recover at the same rate during 1995 and 1996. The timing and magnitude of the market decline cannot be predicted with any accuracy. This assumption merely recognizes that a decline in sales is likely to occur at some time during the forecast period.

Table 4.2

Estimated Retail Sales Of Passenger Cars, 1987 - 1996

Year	Retail Sales
1987	985 000
1988	990 000
1989	995 000
1990	1 000 000
1991	1 005 000
1992	1 010 000
1993	909 000
1994	818 000
1995	909 000
1996	1 010 000

Source: Estimates based on the assumptions listed prior to the table.

Portion of Sales Affected

Some passenger car models sold in Canada are currently equipped to meet United States emission standards and would not require any changes to meet the proposed Canadian standards. These are typically low-volume, top-of-the-line models, such as Corvette, Cadillac, BMW, and Audi. In addition, all American Motors cars produced in North America for sale in Canada and Mazda models imported into Canada are equipped with U. S. federal systems. The automotive manufacturers indicate that the main attractions of offering Canada-specific models, rather than treating Canada and the United States as a single market, are the opportunities to improve fuel economy and to minimize emission control system costs while meeting the Canadian standards. However, the additional costs to develop and certify Canadian emission control systems are usually incurred only if the sales volume of the particular model is large enough to recover the costs. For the 1984 model year, approximately 15 percent of the passenger cars sold in Canada will be fitted with emission control systems that meet United States standards.

As the costs of United States emission systems come down and as the fuel economy of motor vehicles fitted with them improves, the incentive to provide Canada-specific models will decrease. In addition, as long as there is uncertainty as to Canadian emission standards that may be in effect 2 years from now and beyond, some automobile manufacturers have indicated reluctance to develop any new Canada-specific systems. For these reasons, as new car models and engine families are introduced, a higher percentage of passenger cars sold in Canada will be equipped to meet United States emission standards than is currently the case. However, since there is no basis to estimate the magnitude of this likely change and in order to remain consistent with the previously-stated assumptions (that will bias cost estimates upwards), it is assumed that 15 percent of the cars sold in Canada each year between 1987 and 1996 will be equipped to meet United States emission standards regardless of emission standards in effect in Canada during that time period.

Average Incremental Retail Price Equivalent

In response to a survey of the domestic manufacturers and selected importers, industry comments were uniform in stating that a closed-loop, three-way catalyst system would be required to meet the proposed standards --that is, the same systems as currently used on United States models (DeKany and Pilorusso 1983).

Ford, Chrysler, and several of the importers provided price estimates. Ford estimated a retail price equivalent of $175 to replace the current emission control system with one that will meet the proposed standards. Chrysler estimated a $210 incremental retail price equivalent to upgrade its 2.2-litre engine family from an oxidation-catalyst system to a three-way catalyst system. This may understate the incremental retail price equivalent for Chrysler since its 2.2-litre engine family is currently equipped with an emission control system that does not use any catalyst. The incremental retail-price-equivalent estimates provided by importers ranged from $155 (Volkswagen) to $940 (Renault.) The two largest importers, Toyota and Honda, each with about 6 percent of the market, reported

incremental retail price equivalents that generally fall in the $350-$550 range, depending on the engine family and emission control system configuration.

General Motors declined to provide cost or price estimates to upgrade its current emission control systems to meet the proposed standards. However, General Motors stated, "in the interim, we have no objections to you using publicly available industry cost estimates since we expect to be competitive with others at the first cost to the consumer level." Over 90 percent of General Motors passenger cars sold in Canada are equipped with conventional carburetors and oxidation catalysts. General Motors indicated that, to meet the proposed standards, it would probably use its current U. S. federal emission control system with "some refinement anticipated." The major changes from the current systems would be to use carburetors with electronic feedback control and a catalyst consisting of three-way and oxidation catalysts. From the limited EPA cost data that are available, the estimated incremental retail price equivalents would be $30 to $100 and $80 to $140 for the catalyst and the hardware, respectively. Therefore, the incremental retail price equivalent for General Motors passenger cars would be $110 to $240. The midpoint of this range $175 was selected for the calculations which follow.

For the purpose of calculating costs to the consumer, it will be assumed that market shares held by the domestic manufacturers and importers remain at 1983 levels during 1987-1996. The average incremental retail price-equivalent would then be the weighted average of the manufacturers' and importers' estimates discussed above.

The market shares and incremental retail price equivalents for General Motors, Chrysler, Ford, and all others (importers, Volvo, and American Motors) are shown in Table 4.3.

Table 4.3

Average Incremental Retail Price Equivalent

Company	Market Share (%)	Incremental Retail Price Equivalent (1983 $)
General Motors	38	$ 175
Chrysler	16	210
Ford	15	175
All others	31	350
Weighted Average		$ 235

Sources:

Market Share: calculated from sales figures for 1983, published in Ward's Automotive Reports, January 23, 1984.

Incremental Retail Price Equivalents: Chrysler and Ford are the manufacturers' estimates; General Motors and all others are the authors' estimates.

Total Incremental Cost

The total incremental costs for emission control equipment at the manufacturer and retail levels are shown in Table 4.4. The figures are based upon an average of $235 retail price equivalent per vehicle.

Table 4.4

Total Cost of Emission Control Equipment (millions 1983 $)

Year	Number of Cars Affected	Total Incremental Cost		Present Value in 1983	(10% discount)
		Manufacturer	Retail	Manufacturer	Retail
1987	837 000	118	197	81	135
1988	841 000	119	198	74	123
1989	846 000	119	199	67	112
1990	850 000	120	200	62	103
1991	854 000	121	201	56	94
1992	859 000	121	202	52	86
1993	773 000	109	182	42	70
1994	695 000	98	163	34	57
1995	773 000	109	182	35	58
1996	859 000	121	202	35	59
Total				538	897

The present value during 1983 of the total incremental retail price equivalent, discounted at 5, 10, and 15 percent per year, is:

Discount Rate (percent)	Present Value 1983 (millions 1983 $)	
	Manufacturer	Retail
5	773	1 288
10	538	897
15	385	642

Evaporative Emission Standards

United States motor vehicle emissions regulations include a 2-gram sealed housing evaporative determination (SHED) standard. If the proposed Canadian standards include a 2-gram SHED requirement, some models currently sold in Canada may have to be upgraded. Most passenger cars sold in Canada are

equipped with evaporative emission control systems that already meet this standard. Toyota reported that most of its light-duty vehicles already comply, but its 1.5 and 1.6-litre engine families would require modifications costing $23 at the 1983 retail price equivalent level.

It is not known which, if any, other models require upgrading of their evaporative emission control systems to meet a 2-gram SHED standard or what the total costs might be. However, the impact on the retail price would be small in comparison to the total cost estimates shown in Table 4.4.

Diesel-Engined Passenger Cars

Diesel-engine emission control systems that will meet the 0.62 g/km NOx standard as well as diesel particulate standards that are scheduled to come into effect in the United States in the late 1980s are currently being developed. Very little cost/price data for these systems are available; however, the retail price equivalent could be as high as $900.

Diesel-engined passenger cars accounted for about 1.5 percent of total sales in Canada in 1983. Because of the low volume, the incremental retail price equivalent for diesel-engined passenger cars would be small in comparison to the totals shown in Table 4.4. Per vehicle costs could be high, however, if the proposed standards are applied, without change, to diesel-engined passenger cars. This would have a significant impact on Mercedes-Benz since about three quarters (about 2 000 cars per year) of its cars sold in Canada have diesel engines. No other manufacturer uses diesel engines for a large portion of its cars sold in Canada.

Summary - Passenger Cars

Based on the estimate of $235 dollars per car, retail price equivalent, the maximum total cost to passenger car buyers of implementing the proposed standards is approximately $200 million per year. The net present value in 1983 of the maximum additional cost to car buyers over 10 years ending in 1996, discounted at 10 percent per annum, is 897 million (1983) dollars.

The actual cost to car buyers may be less if the costs to the vehicle manufacturers of emission control systems continue to come down and if manufacturers choose not to apply their full, normal markups to their incremental costs for the systems.

4.1.2 Light Trucks

Adoption of the U. S. light-truck emission standards would result in few changes in current emission control systems. For many light-truck models, little or no change would be required. For models that currently use an oxidation-catalyst system, but do not provide sufficient control to meet the standard, the primary change would be to increase the volume and noble metal content of the catalyst. The retail price equivalent of such a change would be about $50. In a few cases the system may have to be upgraded from a noncatalyst to an oxidation-catalyst system.

Complete data on the portion of light trucks that would be affected and the costs involved if U. S. emission standards were adopted are not available. However, the costs would be modest in comparison to the costs to upgrade passenger cars to meet U. S. standards. If we assume that 100,000 of the roughly 250 000 light trucks sold in Canada each year require changes costing $50 retail price equivalent, the total maximum incremental cost to the consumer would be about $5 million (as compared to about $200 million for passenger cars.)

4.1.3 Allocative Costs

If a benefit-cost analysis of a project is to be carried out, the distinction between allocative costs and costs to the consumer becomes a significant one. The portion of the retail price equivalent that is considered to represent allocative costs is a matter of interpretation. According to Treasury Board, only the cost of emission control hardware to vehicle manufacturers and the direct cost to mount it on vehicles fall within the definition of allocative costs (Private Communication, J. Korwin, Treasury Board). The components of the allocative costs are shown in Table 4.5. By this definition, approximately 60 percent of the total retail price equivalent of $ 897 million is the allocative cost, i.e., $538 million.

Table 4.5

Components of Allocative Costs (1983 $)

Component	% of Retail Price Equivalent	Amount per Vehicle
Plant manufacturing cost		
° material	31.4	$ 74
° labour	9.6	24
° plant overhead	3.8	9
Subtotal	44.8	107
Component vendor markup	10.3	24
Tooling expense	2.9	7
Assembly, engine and body modifications	2.0	5
Total--vehicle manufacturer cost	60.0	$143

By the Treasury Board definition the allocative cost is equal to the vehicle manufacturer cost.

4.2 Benefits of Reducing Light Duty Vehicle Emissions Levels

Implementation of the proposed light duty vehicle emission standards would reduce total emissions of hydrocarbons, carbon monoxide, and oxides of nitrogen. The relative contribution of light duty vehicles during 1980 is shown in Table 4.6.

Table 4.6

Nationwide Emissions of Hydrocarbons, Carbon Monoxide, and Nitrogen Oxides By Source, 1980

Sector/Source	Hydrocardons Quantity*	Hydrocardons %	Carbon Monoxide Quantity*	Carbon Monoxide %	Nitrogen Oxides Quantity*	Nitrogen Oxides %
Industrial Processes						
-Iron and Steel			269.4	2.6		
-Petroleum Refining	134.0	6.8	173.6	1.7	5.6	0.3
-Petrochemicals	211.4	10.7				
-Aluminum Manufacturing			149.0	1.4		
-Carbon Black			112.0	1.1		
-Nonferrous Smelters					9.9	0.6
-Tar Sands					24.4	1.4
-Other	226.8	11.5	81.9	0.8	23.0	1.3
Power Generation	5.8	0.3	20.3	0.2	229.4	13.3
Fuel Combustion						
-Petroleum Refining					34.9	2.0
-Natural Gas Processing	45.4	2.3			129.4	7.4
-Other Industrial					104.0	6.0
-Commercial					36.5	2.1
-Residential (incl. fuel wood)			392.7	3.8	50.1	2.9
-Other	15.9	0.8	68.3	0.7		
Transportation						
-Light-Duty Vehicles	460.2	23.2	4719.2	45.1	355.7	20.4
-Medium and Heavy Duty Trucks	140.5	7.1	1716.0	16.4	209.9	12.0
-Off-highway Mobile (air, marine, rail)	86.8	4.4	304.3	2.9	180.1	10.3
-Other**	109.7	5.5	1036.5	9.9	326.2	18.7
Miscellaneous						
-Gas/Diesel Marketing	235.7	11.9				
-Coatings Application	141.6	7.2				
-Solvent Use	76.7	3.9				
-Slash Burning			1040.6	9.9		
-Other	87.9	4.4	385.1	3.7	23.2	1.3
Total	1978	100.0	10469	100.0	1742	100.0

* Thousands of tonnes

** Diesel used by agriculture, forestry and construction equipment; and gasoline used for other purposes, e.g., motorcycles and snowmobiles.

Source: Environmental Protection Programs Directorate, Environment Canada.

The estimated decline in light duty vehicle emissions if the proposed standards are implemented is shown in Table 4.7.

Table 4.7

Projection of Light Duty Vehicle Emissions ('000 tonnes per year)

Substance/ Standard	Year 1990	1995	2000
HC			
- Current	259	259	280
- Proposed	209	137	147
- Reduction (%)	50 (19)	122 (47)	133 (47)
CO			
- Current	4 036	4 272	4 602
- Proposed	3 137	2 061	2 220
- Reduction (%)	899 (22)	2 211 (52)	2 382 (52)
NOx			
- Current	331	357	384
- Proposed	260	196	211
- Reduction (%)	71 (21)	161 (45)	173 (45)

Source: Environmental Protection Programs Directorate, Environment Canada.

The near-term, tangible benefits of reduced light-duty vehicle emission levels are a reduction in the incidence of the adverse effects of (all) automotive emissions shown in Table 3.2. In summary form, the main tangible benefits that may be expected relative to the situation that would prevail if standards are maintained at current levels are:

- o A reduction in the incidence of adverse health effects due to exposure to elevated levels of carbon monoxide (CO), nitrogen dioxide (NO_2), and ozone (O_3).

- o Increased production of forests and agriculture crops in areas where production is reduced due to ambient ozone concentrations.

- o A decrease in the incidence and severity of short-term acidification of freshwater during spring melt of snowpack containing nitrate.

These benefits will be realized in proportion to lower ambient concentrations of CO, NO_2, and O_3 and lower deposition rates of nitrates that result from a reduction in light-duty vehicle emissions. Reductions in ambient HC, CO, and NO_2 concentrations are certain, especially in larger cities. The magnitude of the reductions is shown in Table 4.7. The deposition rate of nitrates is also expected to fall if motor vehicle NOx emissions are reduced.

Reliable, city-specific emission inventory data and appropriately-calibrated u.ban photochemical models needed to estimate the impact of reduced light-duty vehicle HC and NOx on ambient ozone concentration are not available. However, from the discussion in section 2.4.4 it is reasonable to expect that if both HC and NOx emissions are reduced simultaneously O_3 levels will also decrease.

There are other, potentially significant long-term benefits of reducing the amount of HC, NO_x and CO emissions that are released into the atmosphere. The dose-response relationships between ambient concentrations of pollutants and the probability of an adverse effect are not well enough understood to allow estimation of precise probabilities. The science of atmospheric photochemistry is not sufficiently well developed to predict with a high degree of certainty the relationship between primary and secondary pollutants, e.g., the precise contribution of HC and NO_x to O_3 formation. (The secondary substance, O_3, is a more significant air quality problem than the primary substances, HC and NO_x). There is a probability that long-range transport of air pollution

caused by motor vehicle emissions from urban areas contributes adverse effects in regions some distance from their source.

Reducing light-duty vehicle emissions appears to be a necessary, but not necessarily sufficient, condition to attain any desired reduction in ambient ozone concentrations. In some cities, reductions in light-duty vehicle HC and NOx emissions will result in lower ambient ozone concentrations. However, in most cities, reductions in HC emissions from other sources, e.g., gasoline marketing and application of surface coatings, will also be required in order to achieve ambient air quality objectives for ozone.

Under most assumptions of population and economic growth, total emissions from all sources of HC, CO, and NO_x will increase toward the end of the century unless specific action is taken to control them. As discussed in greater detail in section 4.2.1, total light duty vehicle emissions are not expected to increase significantly over current amounts if the current emission standards are retained.

4.2.1 Allocative Benefits

Benefit-cost and cost-effectiveness analyses are the two methodologies suggested by Treasury Board (1979) as being appropriate for dealing with the allocative effects of proposed government actions.

Benefit-cost analysis is a systematic attempt to identify and measure all relevant social costs and benefits in monetary terms. The dose-response functions and air quality projections over the years 1987 to 1996 required to calculate dollar values for allocative benefits are not available. It is, therefore, not possible to make a comparison of the monetary value of benefits and the costs of implementing the proposed standards for the purpose of assessing the economic efficiency of the proposed emission standards.

When it is not possible to measure the benefits of a proposed action in dollar terms, cost-effectiveness analysis can be used to determine the minimum total

cost of meeting a given objective. This is done by examining a number of alternatives, in terms of their costs, for achieving a specific objective. In the current case the objective could be stated in terms of the number of tonnes by which emissions of HC, NO_x, and CO are to be reduced. We have calculated the costs to meet the objective by reducing light duty vehicle emissions. These costs are compared to the costs of alternative means to reduce emissions of HC, CO, and NOx.

Exceedances of National Ambient Air Quality Objectives[1]

Exceedances of the National Ambient Air Quality Objectives for CO in Toronto, Montreal, Vancouver, and Edmonton during 1979, 1980, and 1981 are shown in Table 4.8

Table 4.8

Exceedances of National Ambient Air Quality Objectives For CO

City	Number Greater Than 1-Hour Max Desirable*			1-Hour Max Acceptable**		
	1979	1980	1981	1979	1980	1981
Toronto	8	0	2	0	0	0
Montreal	85	44	37	0	0	0
Vancouver	35	10	52	0	0	0
Edmonton	124	68	79	0	0	0

* 15 mg/m^3 (13 ppm)

** 35 mg/m^3 (31 ppm)

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1 Air quality data presented here are from tabulations in Davis et al (1984).

There are specific areas within the cities where concentrations are higher than those shown in Tables 4.8, 4.9, and 4.10. For example, no exceedances of the 1-hour maximum acceptable objective for CO are shown. However, as noted in section 2.4.1, during 1980, a maximum 8-hour average of 29.9 mg/m (26 ppm) of CO and a 1-hour maximum of 43.7 mg/m (38 ppm) of CO were recorded at 381 Yonge Street in Toronto. If people were exposed at these levels for the full 1 hour or 8-hour period during the air pollution episodes, some portion of them could be expected to experience adverse health effects.

On any given day there is a certain probability that some areas within the cities will have concentrations of CO above the NAAQO acceptable limit. The higher the city-average CO concentration, the higher will be the probability of exceedances in certain areas of the city. During the time that an area experiences a CO episode (i.e., concentrations above the NAAQO acceptable limit) there is a second probability that members of populations that are susceptible to adverse effects from CO will be in the area and thus exposed to risk. If the density functions for the two probabilities for each area in which CO episodes occur were known, it would be possible to estimate the incidence of the health effects listed in Table 3.2. The probability density functions are not known and it is not possible to estimate the incidence of adverse effects associated with ambient CO concentrations.

Exceedances of National Air Quality Objectives for nitrogen dioxide (NO_2) during 1979, 1980, and 1981 for the four cities are shown in Table 4.9.

Table 4.9

Exceedances of National Ambient Air Quality Objectives For NO_2

City	Number Greater Than 1-Hour Max Acceptable*			1-Hour Max Tolerable**		
	1979	1980	1981	1979	1980	1981
Toronto	0	1	0	0	0	0
Montreal	0	0	5	0	0	0
Vancouver	1	0	0	0	0	0
Edmonton	0	2	1	0	0	0

* 400 $\mu g/m^3$ (213 ppb)

** 1000 $\mu g/m^3$ (532 ppb)

The comments on CO made above are also applicable to NO_2. Some adverse health effects at current ambient levels of NO_2 are possible, but the extent to which they may be occurring is not known.

Exceedances of National Air Quality Objectives for ozone during 1979, 1980, and 1981 for the four cities are shown on Table 4.10.

Table 4.10

Exceedances of National Ambient Air Quality Objectives For Ozone

City	Number Greater Than 1-Hour Max Desirable*			1-Hour Max Acceptable**		
	1979	1980	1981	1979	1980	1981
Toronto	1231	1199	1177	177	156	131
Montreal	1909	1109	1326	262	151	208
Vancouver	1591	1694	1621	208	279	459
Edmonton	190	392	87	0	32	4

* 100 $\mu g/m^3$ (50 ppb)

** 160 $\mu g/m^3$ (80 ppb)

Although the adverse health effect threshold concentration for ozone is unknown, there is an increased probability of adverse health effects when the ambient ozone concentration reaches 200-300 g/m^3 (100-200 ppb) (See Table 3.2). Therefore, the number of exceedances of the 1-hour maximum acceptable NAAQO, 160 $\mu g/m^3$ (80 ppb), shown in Table 4.10, indicates that current ambient concentrations of ozone may be a threat to human health. For the months of July and August, Bates and Sizto (1983) found statistically significant associations between excess hospital admissions for respiratory conditions and ambient concentrations of sulfur dioxide, ambient concentration of ozone, and temperature in southern Ontario.

Populations at Risk[2]

The size of the populations at risk to the deleterious effects of air pollution in four major cities and all of Canada are shown in Table 4.11.

Table 4.11

Size of Populations at Risk, 1981

Population at Risk	City				All Canada
	Edmonton	Montreal	Toronto	Vancouver	
1 General Population	657 057	2 828 349	2 998 947	1 268 183	24 343 180
2 Children (under 15)	73 280	279 850	307 230	120 430	5 481 105
3 Population with cardiovascular disease	16 870	63 940	77 560	33 690	606 170
4 Population with acute respiratory disease	22 340	96 160	101 960	43 120	827 668
5 Bronchitics and emphysematics	15 770	67 880	71 970	30 440	584 236
6 Asthmatics	15 770	67 880	71 970	30 440	584 236
7 Anemics	11 830	50 910	53 980	22 830	438 177
8 Pregnant women	6 804	22 115	23 933	11 015	203 678
9 Newly-born infants	12 550	41 860	42 590	19 150	372 451

During a given air pollution episode not all of the population at risk will be exposed for a sufficient length of time at a high enough concentration to be adversely affected. Most Canadians spend most of their time indoors. Concentrations of nonreactive gases such as carbon monoxide do not vary as much indoors as outdoors when no major indoor sources are present. The overall indoor concentration is equal to the average outdoor concentration, but peak indoor concentrations of CO are usually less than peak outdoor concentrations. Concentrations of reactive gases such as ozone and nitrogen dioxide do not vary as much indoors as outdoors in the absence of major indoor sources. The average indoor concentrations of reactive gases are less than

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2 The discussion of populations at risk draws extensively from Pilorusso et al (1984).

outdoors. Therefore, outdoor concentrations of CO, NO_2, and ozone would have to rise above the minimum outdoor levels at which adverse effects are observed before large portions of the populations at risk (i.e., the portion that is indoors) are affected.

The dose of a substance that an exposed person receives is determined by pollutant concentration, rate of ingestion, and duration of exposure. Rate of ingestion is proportional to ventilation rate, which is dependent on the specific activity being performed by an individual.

By far the greatest amount of an individual's time is spent in the home--about 14.5 hours of the day. On average, about 7.5 hours of that time are spent resting during which ventilation rates are the lowest and, therefore, ingestion rates are reduced. People who work outside of the home typically spend 35 to 40 hours per week in their workplace. For indoor workers, air quality is determined by the same factors as residences. However, indoor sources of pollutants may be a much greater factor in the workplace than in the home. Approximately 1.6 hours per day are spent in closed vehicles. Air quality in this environment is dependent primarily on type and condition of vehicle, traffic volume and speed, and the presence of smokers. Finally, on average, 1.2 hours per day are spent out of doors. Often, outside activities (e.g., sports and physical activities) are associated with increased ventilation rates (e.g., twice normal rates) so that effective dosage outdoors may be proportionately greater than the estimate of the amount of time spent out of doors may suggest. Taking this fact into consideration, however, will not significantly affect the dominance of indoor air quality in determining cumulative exposure rates.

For the reasons outlined above and in the preceding discussion on exceedances of National Ambient Air Quality Objectives, not all, but a fraction, of the populations at risk listed in Table 4.11 are exposed to the risk of experiencing an adverse health effect from air pollution. Of those exposed, not all, only the most susceptible will experience an adverse health effect.

In order to determine the portion of the population that may be exposed to

ambient concentrations of pollutants for a long enough period to experience an adverse effect, two types of data are required. The first is a fairly extensive data base of ambient pollutant concentrations at street level for a number of typical urban areas. The air quality data presented in Tables 4.8, 4.9, and 4.10 are based on data collected from monitors at various heights located above street level and may not be representative of the air that people on the street breathe. The locations of the monitors were chosen so that trends in air pollution from all sources could be monitored.

The second type of data that is required is human exposure data from which it would be possible to estimate the amount of time individuals spend in areas in which exceedances of ambient air quality objectives are known to occur.

Neither type of data are available for Canada. In Sweden, where air quality data from monitors located 3 metres above street level are available, it was found that in cities with populations greater than 10,000, approximately 2 percent of the population live in areas where the WHO guidelines for carbon monoxide and nitrogen dioxide levels are exceeded. (Swedish Government 1983). Another 2 percent of the population work in streets where the guidelines are exceeded, although some of these are included in the previous 2 percent. Care must be taken in inferring too much from the Swedish experience since motor vehicle emission standards, other sources of pollutants, population densities, meteorology, and other factors that can affect ambient pollutant concentrations and exposure are different in Canada than in Sweden.

The Costs of Adverse Health Effects[3]

The economic evaluation of health impacts is generally approached in two main ways. These can be referred to as:

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3 The cost estimates were prepared by P. A. Victor and included in Pilorusso et al (1984).

- output accounting: the value of the output lost and resources foregone because of death, illness, or injury.

- welfare accounting: the value of resources individuals would forego (accept) to reduce (increase) a risk of death, illness, or injury if they had the opportunity.

It is generally considered that welfare accounting is the more theoretically-sound approach. But practical problems limit its usefulness in evaluation of health impacts. The first is that only a small number of empirical studies using a welfare accounting approach have been published. Only one Canadian study using the welfare account methodology has been identified.

The second practical difficulty is that most of the welfare accounting studies deal with mortality, whereas morbidity is the central concern with respect to automotive emissions. Despite the theoretical appeal of welfare accounting, it has been necessary to adopt an output accounting framework to evaluate health effects.

Table 4.12 shows some preliminary estimates of the per-case costs of medical treatment of the diseases that could be caused by excess exposure to CO, NO_x, and ozone. The figures shown in the table include estimates for direct hospital costs (for cases where hospital treatment is required), physicians' costs, prescription drug costs, and an amount for diagnostic services.

The other categories of costs that can be estimated with some accuracy but are not included in the figures shown in Table 4.12 are: the value of lost time in paid employment and the value of lost household work. The aggregate of these costs and medical costs will not be as large as welfare accounting values since they do not include intangibles such as the value of pain and suffering. However, the output accounting estimates can be interpreted as a lower bound on the welfare accounting values.

The next step in quantifying health effects would be to estimate the decline

Table 4.12

Estimated Medical Costs
($1983)

Substance/Effect	ICD9 Code	Hospital Treatment cost/case		Nonhospital Treatment cost/case	
		Male	Female	Male	Female
Carbon Monoxide					
1. Myocardial infarction (heart attack)	410	1736	1709	63	56
2. Aggravation of angina pectoris (attacks of sharp chest pain due to inadequate blood supply)	413	1400	1385	64	59
3. Intermittent claudication (episodes of leg pain or limping due to inadequate blood supply)	411	1305	1322	58	68
4. Impaired vigilance (visual, auditory, reaction time, time estimation, etc.)	781	1113	1109	45	44
5. Effect on fetal development (decreased birth weight, increased perinatal deaths and decreased weight gain in newborns; reduced learned and general abilities in later childhood)	319	1650	1768	76	103
	635	na	1070	na	115
	646	na	889	na	36
	647	na	1490	na	*
	648	na	1619	na	72
	653	na	1559	na	102
	654	na	1562	na	67

Table 4.12 continued

Estimated Medical Costs
($1983)

Substance/Effect	ICD9 Code	Hospital Treatment Cost/Case		Nonhospital Treatment Cost/Case	
		Male	Female	Male	Female
Nitrogen Dioxide					
6. Increased incidence of acute respiratory illness	460	966	961	34	34
	466	1001	995	41	40
7. Aggravation of respiratory disease (attacks of asthma, bronchitis, or emphysema)	490	1266	1253	42	41
	491	1279	1266	52	46
	492	1317	1305	62	65
	493	1260	1273	87	86
8. Impaired lung function and likelihood of incidence of respiratory symptoms (chest discomfort, cough, shortness of breath)	443	1318	1278	54	47
	478	2023	2014	40	38
Ozone					
9. Increased incidence of discomfort and respiratory symptoms (chest discomfort, cough, headache, sore throat, eye irritation, shortness of breath)	379	1641	1612	47	47
	443	1318	1278	54	47
	478	2023	2014	40	38
10. Aggravation of respiratory disease (same as NO_2)	490	1266	1253	42	41
	491	1279	1266	52	46
	492	1317	1305	62	65
	493	1260	1273	87	86
11. Decreased resistance to infection	460	966	961	34	34
	466	1001	995	41	40
	480	1532	1531	43	38
	487	1004	1009	38	38

na - not applicable

* - no data.

in the number of cases of diseases that may result if the proposed light-duty vehicle emission standards are implemented. Two types of data are required to make this estimate. The first is dose-response functions for each adverse effect. Dose-response functions are the mathematical relationship between the dose of a pollutant that an individual receives and the probability that he will experience a particular adverse effect.

The second type of data are estimates of the impact of the proposed light-duty vehicle standards on the number, ambient concentrations, and duration of exceedances of National Air Quality Objectives for CO, NO_x, and ozone.

Agricultural Crops

There are two crop-growing areas in Canada where mean, growing-season ozone concentrations are above the threshold for agricultural crop damage: (1) the southwestern portion of Ontario and Quebec along the Windsor to Quebec City corridor; and (2) Vancouver and the lower Fraser River Valley. Southwestern Ontario and Quebec are most affected by long range transport of ozone, with ozone transport from the United States strongly implicated (EPS 1984). Elevated concentrations of ozone occur in the regions downwind of Toronto and Montreal (Mukammal et al 1982, Vukovich et al 1977). For Vancouver and the lower Fraser River Valley emissions from Vancouver are the main contributor to the ozone problem (Concord 1982).

The extent to which the mean, growing-season ozone concentration would fall if the proposed light duty vehicle emission standards are implemented is not known. Determining the impact of light duty vehicle emissions on ambient ozone concentration is particularly difficult in areas such as Southwestern Ontario where long range transport of ozone plays a significant role. It has been estimated that crop losses in Ontario due to ozone damage average about $15 million per year, with a range of $9 million to $23 million (Linzon et al 1984). There is at present no methodology for estimating the portion of crop losses in Ontario that can be attributed to light duty vehicle emissions.

The total values of agricultural crops that are at risk of ozone damage in the Fraser River valley and Quebec are: $37 million and $566 million respectively (Statistics Canada 1984a). Estimates of total ozone damage or the amount that can be attributed to light duty vehicle emissions for these regions are not available.

Forests

Losses in forest productivity due to elevated concentrations of ozone are potentially large. However, there are few data available for either ambient O_3 concentration in nonurban areas or detailed losses of forest productivity. Therefore, it is not possible to either derive dose-response relationships or identify, with any precision, the areas of forest at risk.

Until the phenomenon of long-range transport of pollutants is better understood and data on ambient concentration of O_3 in forested areas are available, it is not possible to determine the value of forest-productivity losses that may be due to elevated levels of ozone.

The total value of the annual forest growth of 150 241 000 m^3 in eastern Canada is estimated to be $3.9 billion, based upon an average wood value of $26.25 per m^3 in 1981 (MOI 1983).

Short-Term Freshwater Acidification

The contribution of nitric acid to short-term freshwater acidification and the consequent risk to aquatic species during spring melt of snowpack containing nitrate has been identified in the literature and discussed briefly in Section 3.2. The information required to quantify the effect such as the number and size of freshwater bodies affected, the change in acidity, the types and number of organisms at risk, and dose-response functions are not yet available. Mathematical models are not available at this time to determine the relative contribution of nitrates from light duty vehicles to total

nitrate accumulation in the snowpack over the winter. An approximation of the light-duty vehicle contribution can be made. Galloway and Dillon (1983) estimated that in northeastern United States and Canada $SO_4^=$ contributes 65 percent to the precipitation acidity while NO_3^- contributes most of the balance. If we assume that light-duty vehicle NOx emissions contribute to precipitation acidity in the same proportion as other sources of NOx, the light-duty vehicle contribution to precipitation acidity is approximately 7 percent (35 x 0.2). If the further assumption of a linear relationship between NOx emissions and precipitation acidity is made, the light-duty vehicle contribution to acidic precipitation will fall from 7 percent to about 4 percent (7 x 0.55) if the proposed standards are implemented, i.e., an anticipated reduction of approximately 45 percent of light duty vehicle NOx emissions.

Man-Made Materials

While the adverse effects of NO_2 on fabrics--dye fading for a number of fabric and dye combinations and yellowing of some white fabrics--have been well established, the degree to which economic losses occur due to exposure to NO_2 is unclear. The two main problems in quantifying these effects are: the laboratory studies demonstrating the effects are not representative of real-life exposure; and the effects of NO_2 have not yet been successfully separated from the effects of temperature, humidity, and other pollutants.

Evidence exists to show that nitrogen can affect the rate of oxidation of metals and alloys, but a direct relationship between NO_x or NO_3^- levels in ambient air and corrosion behaviour has not been established. Therefore, quantifying this effect is not possible without further research.

The economic cost of ozone damage to tires is made up mainly of the cost of the antiozonants that are added to sidewall rubber compounds. To the extent that lower levels of light-duty vehicle emissions would lead to lower ambient ozone levels and, in turn, to the use of less antiozonants, the dollar value

of this effect can be determined. However, rubber compounds are formulated for worst case situations. For tires manufactured in Canada the worst case would be the United States cities with the highest ozone concentrations. Since changes in Canadian light-duty vehicle emissions levels will have no effect on the worst case situation, Canadian tire manufacturers are unlikely to reduce the amounts of antiozonants used in the tires they produce.

Visibility

Two approaches are generally used for economic valuation of visibility: property value studies and surveys using iterative bidding techniques. The only approach in which it is possible to separate the value of improved visibility from other benefits of reduced air pollution, such as reduced risk to health, is the iterative bidding methodology.

The number of published iterative bidding studies that attempt to evaluate visibility is very small. None is known to have been carried out in Canada, and only about a half dozen such studies have been carried out in various parts of the United States. Although they are not particularly relevant for the situation in Canada, the United States studies indicated that--depending on the area considered, the expected improvement in visibility, and a number of other factors--the willingness-to-pay estimates for individual residents ranged from $43 to $245 per year.

Impact of the Proposed Emissions Standards on Air Quality

The difference between expected air quality if no action on light-duty vehicle emission standards is taken (the baseline) and if the proposed emission standards are implemented is the measure of the impact of the proposed standards considered here. The impact would first be apparent in the year in which the proposed standards first came into effect. It would take approximately 10 years before the entire light-duty vehicle fleet was equipped to meet the proposed standards. Therefore, if the proposed standards come

into effect in 1987, it would be 1996 or 1997 before their full impact on air quality would be apparent. These air quality benefits would continue after 1997 for as long as the proposed emission standards remain in force.

Ideally, the impact of the proposed emission standards would be measured in terms of changes in the ambient concentration of carbon monoxide, nitrogen dioxide, and ozone, and the change in the nitric acid levels of acidic precipitation. However, reliable estimates of these changes cannot be made from the data available for Canada. Consequently the discussion of the impact of the proposed emission standards is effectively limited to considering the reductions in the primary motor vehicle emissions--HC, CO, and NO_x that are emitted to the atmosphere.

Projected emission factors for the in-use light-duty vehicle fleet are shown in Table 4.13. The emission factors shown in Table 4.13 tend to be higher than new vehicle standards since they take into account such factors as vehicle age distribution and increases in emissions due to deterioration of, or tampering with, emission control equipment.

Table 4.13

Projected Vehicle Emission Factors

Vehicle Type	1980			1985			1990			1995		
	HC	CO	NOx	HC	CO	NOx	HC	CO	NOx	HC	CO	NOx
Passenger Cars												
g/km	2.35	22.51	1.74	1.44	17.89	1.50	1.11	18.09	1.44	1.01	17.72	1.44
(g/mile)	3.78	36.22	2.80	2.32	28.79	2.42	1.78	29.11	2.31	1.63	28.52	2.31
Light Trucks												
g/km	2.47	34.16	2.33	1.60	19.71	1.88	1.37	16.74	1.65	1.37	16.74	1.65
(g/mile)	3.97	54.97	3.75	2.58	31.72	3.03	2.21	26.94	2.65	2.21	26.94	2.65
Composite LDV												
g/km	2.36	23.68	1.80	1.46	18.07	1.54	1.13	17.95	1.45	1.05	17.62	1.45
(g/mile)	3.80	38.10	2.90	2.35	29.08	2.48	1.82	28.89	2.34	1.69	28.36	2.34

Source: Passenger Car and light-truck emission factors: Transportation Systems Division, Industrial Programme Branch, Environment Canada as published in Davis et al (1984).

Composite LDV emission factors are calculated based on passenger cars accounting for 90 percent of light duty vehicle fleet vehicle kilometres travelled.

Vehicle emission factors are expected to decline between 1980 and 1990 for a number of reasons, amongst which are:

- Reduced deterioration rates as emission control technology improves. After light duty vehicles have been in use for a period of time, emissions increase from new-vehicle levels. Later model vehicles can be expected to maintain emissions rates closer to design levels than older model vehicles.

- Increased use of U.S. federal systems on Canadian cars for the reasons stated in section 4.1.1.

- Reduced per-vehicle emissions due to use of catalyst systems for models that currently use noncatalyst systems. The use of noncatalyst systems generally result in poorer fuel economy but allow the use of less expensive leaded gasoline. Now that Canada's fuel prices have reached virtual world levels there is a higher priority on fuel economy and less incentive for noncatalyst systems. Although it cannot be confirmed, some of the manufacturers currently offering non-catalyst models are expected to switch to catalyst technology by the 1987 model year.

- By 1987, the majority of the remaining, high-emitting, pre-1975 model year vehicles will be off the road.

- By 1987, a higher portion of vehicles will be equipped with fuel injection systems in place of carburetors. Fuel-injected engines emit less CO and HC.

Any increases in the light duty vehicle fleet's total vehicle kilometres travelled (VKmT) will offset the declining trend in emission factors. For example, if the compound annual growth in VKmT exceeds 3 percent between 1985 and 1995 the number of tonnes of HC emissions from light-duty vehicles will be higher in 1995 than in 1985. The comparable growth rates for CO and NO_x are 0.25 percent and 0.6 percent respectively. Estimates of the annual growth of light duty vehicle fleet VKmT generally fall in the 0.5 to 1.5 percent range.

Therefore, if the current standards are maintained, increases in light-duty vehicle CO and NO_x emissions and a decrease in HC emissions from light-duty vehicles can be expected. After 1995, Environment Canada is projecting increases in light-duty vehicle HC, CO, and NO_x emissions (see Table 4.7).

4.3 Costs and Benefits of Decreased Demand for Leaded Gasoline

Unleaded gasoline first became available in Canada in 1971, but did not account for more than about 1 percent of total motor vehicle gasoline sales until 1975 when the current light-duty vehicle emission standards first came into effect. Since 1975, an increasing portion of the light-duty vehicle fleet has been equipped with catalyst emission control systems that require unleaded gasoline. The portion of motor vehicle gasoline sales accounted for by unleaded gasoline has increased by about 4 percent per year since 1975. The leaded/unleaded gasoline split is currently about 50/50. If the current emission standards are maintained and manufacturers continue to produce the same relative number of lead-tolerant light-duty vehicles as currently, demand for unleaded fuel is expected to increase by 2 to 3 percent per year until 1990 when an equilibrium grade split of 30 percent leaded, 70 percent unleaded is reached (Hickling 1984).

Annual consumption of gasoline by motor vehicles in the years 1990 and beyond is expected to be between 30 and 35 billion litres. If current emission standards are maintained, virtually all of the unleaded gasoline will be consumed by light-duty vehicles; a further 20 percent of the total will be leaded gasoline consumed by light-duty vehicles; and the remaining 10 percent, also leaded, will be accounted for by heavy trucks and buses and by misfuelling of light-duty vehicles.

If the proposed emission standards are implemented, all light-duty vehicles would require unleaded gasoline. By 1996 the grade split would be 10 percent leaded, 90 percent unleaded. The main allocative impacts of the reduced use of leaded fuel are: a reduction in the risk of adverse health effects due to a decrease in lead emissions; increased costs to refiners to shift production from leaded to unleaded gasoline; and improved fuel economy and lower maintenance costs for the light-duty vehicle fleet.

The issues related to phasing out the use of lead in gasoline have been fully discussed in Socio-Economic Impact Analysis of Lead Phase-down Control Options, prepared by James F. Hickling Management Consultants Ltd., and released by Environment Canada in February, 1984. Only a brief

summary of the major benefits and costs will be presented here.

4.3.1 Health Effects

The issue of lead emissions and adverse health effects was summarized as follows:

"Health and Welfare Canada has determined from recent studies that adverse health effects can occur at blood lead levels of 20-30 g/dL. Furthermore, these studies indicate that:

- significant numbers of Canadian children could have blood lead levels in this range and could therefore be at risk; and
- gasoline lead contributes an estimated 30-35% of the lead in the blood of urban adults. The proportion of gasoline lead in children's blood may range from 30-40%.

Health and Welfare Canada has therefore concluded that a reduction of automotive lead emissions should significantly decrease lead uptake and, consequently, the blood lead levels in Canadians." (Hickling 1984)

To reduce combustion chamber deposits of lead in engines burning leaded gasoline, ethylene dibromide (EDB) and ethylene dichloride (EDC) are added to the gasoline to scavenge the lead. Removing lead from gasoline would have the effect of removing EDB and EDC since the lead scavengers would no longer be required. The resulting decrease of EDB and EDC emissions can also be considered a health benefit.

4.3.2 Benefits and Costs

The benefits and costs of moving from a 30/70 to a 10/90 leaded/unleaded gasoline grade split that can be expressed in dollars are summarized in Table 4.14. Three estimates of costs are shown in Table 4.14. The first are based

upon estimates, provided by the Petroleum Association for Conservation of the Canadian Environment (PACE), of up to $560 million in refinery capital costs and up to $80 million in annual operating costs in 1984 dollars. These estimates include the costs to discontinue the use of methylcyclopentadienyl manganese tricarbonyl (MMT) in unleaded gasoline.

The cost figures in the second and third columns of Table 4.14 are based upon estimates prepared for Environment Canada by Monenco (1984). The figures in the second column include the cost to discontinue the use of MMT. If the use of MMT is to be discontinued, Monenco (1984) estimates that refinery capital costs would be $494 million and operating costs $42 million. The figures in the third column of Table 4.14 do not include the cost to discontinue the use of MMT. If the use of MMT is continued, Monenco (1984) estimates that refinery capital costs would be $68 million and annual operating costs $19 million. All of the Monenco estimates are in 1984 dollars.

To calculate the figures shown in Table 4.14 it was assumed for all cases that capital costs were evenly spread over 10 years starting in 1986 and that operating expenses increased each year starting in 1987 by 10 percent of the operating costs of the tenth year when they reach their maximum value.

MMT is used currently as an octane booster in regular and premium unleaded gasoline sold in Canada. Judging from the experience of the Environmental Protection Agency in the United States and the comments of the automobile manufacturers recorded in DeKany and Pilorusso (1983), there is concern that the proposed hydrocarbon (HC) standard of .25 grams/kilometre cannot be met with a reasonable margin if MMT is used in unleaded fuels.

Other concerns expressed by manufacturers with respect to operating closed loop, three-way catalyst systems on gasoline containing MMT include: plugging of catalysts, premature failure of oxygen sensors and higher combustion chamber deposits. For these reasons and the effect of MMT on HC emissions, the prevailing sentiment of the domestic manufacturers is that, if the proposed emission standards are implemented, steps should be taken to remove MMT from unleaded gasoline.

However, since closed loop, three-way catalyst systems were introduced during the 1981 model year, in the neighborhood of 200 000 passenger cars equipped with the systems have been sold in Canada. Over the last 3 years these vehicles have accumulated a substantial number of vehicle kilometres travelled, while operating on gasoline containing MMT, without apparent, serious problems. Therefore, it is by no means certain that closed loop three-way catalyst systems are incompatible with unleaded gasoline containing MMT. Estimates of the costs to produce additional unleaded gasoline both with and without MMT have been included in Table 4.14.

Table 4.14

Benefits and Costs of Moving From a 30/70 to 10/90 Leaded/Unleaded Gasoline Split, 1987 - 1996 (millions 1983 dollars)

Present Value in 1983 (10% discount)

	PACE No MMT	Monenco No MMT	Monenco with MMT
Costs			
- Refinery Investment	284	251	35
- Refinery Operating	175	92	41
Total	459	343	76
Benefits*			
- Fuel Savings	75	75	75
- Maintenance Saving	68	68	68
Total	143	143	143

* Benefits are based on the retail sales projections shown in Table 4.2 and 20 percent of passenger cars shifting from leaded to unleaded gasoline. Other assumptions are the same as Hickling (1984). They are:

- average distance travelled per car per year: 16 000 km.
- average fuel consumption: 11.7 km/L
- fuel consumption improvement with unleaded gasoline: 2.8%
- average cost of gasoline: $ .46/L

- maintenance saving using unleaded gasoline: $ 1/1 000 km.

The figures shown in Table 4.14 were calculated using a 10 percent discount factor. Estimates of the 1983 present value of costs and benefits using a 5, 10, and 15 percent discount factor are shown in Table 4.15.

Table 4.15

Benefits and Costs of Producing
Additional Unleaded Gasoline
Between 1987 and 1996
(millions 1983 dollars)

	Present Value in 1983		
	5% D.F.	10% D.F.	15% D.F.
Costs			
PACE, No MMT	664	459	328
Monenco, No MMT	499	343	248
Monenco, With MMT	112	76	53
Benefits	224	143	95

4.3.3 Costs to The Consumer

The maintenance and fuel economy savings available to the consumer by switching from leaded to unleaded gasoline are offset by the additional cost of unleaded gasoline. The average difference in the price of regular leaded gasoline and regular unleaded gasoline in Canada is $0.024 per litre (Private communication, D. Young, Energy, Mines and Resources Canada). Using the same assumptions for the average distance travelled per car per year (16 000 km) and average fuel consumption (11.7 km/l) as used previously, the additional cost for unleaded gasoline would be $32.82 per car per year. The savings in maintenance costs and improved fuel economy add up to $33.60 per car per year. Therefore, virtually the entire maintenance and fuel savings are offset by the higher price of unleaded gasoline.

Based on the Monenco estimates and assuming a cost of capital of 15 percent per year if the use of MMT in unleaded gasoline is discontinued the additional

costs to the refiners to produce unleaded gasoline at market octanes will be about $0.0025 per litre. To get to the retail price equivalent, the following must be added to the refiners' cost: refinery, distributor and retail margins; the federal sales tax; and, in all provinces except Alberta and Saskatchewan, the provincial sales tax. These charges are equal to between 40 and 100 percent, depending on the province of the additional cost to the refiners. Therefore, the retail price equivalent of the cost to the refiners to discontinue the use of MMT in unleaded gasoline would be between one-third and one-half cent per litre. On average this would represent an additional cost to consumers of about 5 to 6 dollars per vehicle per year.

4.4 Summary of Benefits and Costs

The expected benefits and costs if the proposed light-duty vehicle emission standards are implemented are summarized in Table 4.16.

Table 4.16

Summary of Benefits and Costs Of Implementing The Proposed Light-Duty Vehicle Emission Standards (1987 - 1996)

			Value * (millions 1983 dollars)
Benefits			
1.	Reduction in the incidence of adverse health effects from exposure to ambient concentrations of CO, NO_2, O_3, and lead.		**
2.	Increased production of forests and agricultural crops in areas where reduced production due to ozone concentrations is observed.		**
3.	A decrease in the contribution of nitrates to precipitation acidity.		**
4.	Intangible and unknown benefits of reduced HC, CO, and NOx emissions.		**
5.	Improved fuel economy due to increased use of unleaded fuel.		75
6.	Lower maintenance costs due to increased use of unleaded fuel.		68
Costs			
1.	Cost of emission control equipment		
	(a) Allocative costs		
	Cost to manufacturer	538	
	(b) Nonallocative costs		
	Vehicle manufacturer markup	124	
	Federal sales tax	59	
	Dealer markup	124	
	Research and development	51	
	Subtotal	358	
	Total cost to the consumer		896
2.	Cost to produce additional unleaded gasoline		
	(a) Refinery capital and operating costs		76 - 459***
	(b) Markups and taxes on the increased cost to refiners		**

* Net present value in 1983 at a discount rate of 10 percent.

** Dollar values could not be established.

*** The estimate of $76 million does not include the costs to remove MMT from unleaded gasoline. The estimate of $459 million is the higher of the two available estimates that include costs to remove MMT.

4.5 Comparison of Costs and Emission Reductions

4.5.1 Proposed Emission Standards

The cost of emission control hardware is incurred in the year that the vehicle is built. The benefits of lower emissions from the vehicle continue for the useful life of the vehicle or, about 10 years.

Assuming that the proposed emission standards come into force in 1987, the costs and the reduction in emissions would be as shown in Table 4.17.

Table 4.17

Costs and Emission Reductions Of Proposed Standards From 1986 Levels

	Costs (millions of 1983 $)*					Emission Reductions (thousands of tonnes)		
	Manufacturer	Retail	Refinery**	Fuel	Maintenance	HC	CO	NO_x
1987	118	197	9	(4)	(3)	12	225	18
1988	119	198	11	(7)	(6)	25	450	36
1989	119	199	13	(11)	(10)	37	675	54
1990	120	200	14	(14)	(13)	50	899	71
1991	121	201	16	(18)	(16)	64	1161	89
1992	121	202	18	(21)	(19)	79	1424	107
1993	109	182	20	(24)	(22)	93	1686	125
1994	98	163	22	(27)	(25)	108	1949	143
1995	109	182	24	(30)	(28)	122	2211	161
1996	121	202	26	(34)	(31)	124	2245	163

* figures in brackets are savings

** based upon the Monenco estimates to produce additional unleaded gasoline containing MMT.

Sources: Costs of the emission control hardware at the manufacturer and retail level are from Table 4.4.
Refinery, fuel, and maintenance costs (savings) are calculated from the figures in Table 4.14.
Emission reductions for 1990 and 1995 are from Table 4.7; reductions for other years are interpolated.

After 1996, costs and emission reductions reach equilibrium. The annual emission reductions of approximately 125 000 tonnes of HC, 2 250 000 tonnes of CO, and 165 000 tonnes of NOx will continue to accumulate over the life of the affected fleet.

If we make the simplifying assumption that the costs can be equally divided amongst control of HC, CO, and NO_x, the approximate costs at the retail level per tonne in constant 1983 dollars for emission reductions are: $432/tonne of HC, $26/tonne of CO and $355/tonne of NOx. If only the allocative costs of the emission control hardware are considered the figures become: $240/tonne of HC, $14/tonne of CO and $198/tonne of NO_x. All of the forgoing estimates include the refinery costs to produce the additional unleaded fuel that would be required, but not the costs to discontinue the use of MMT. If the costs to discontinue the use of MMT are included it will add approximately $137/tonne for HC, $8/tonne for CO, and $113/tonne for NO_x.

In order for the cost/tonne estimates presented in the previous paragraph to be used in a cost-effectiveness comparison, two conditions must be met. The first is that the costs/tonne estimates of alternative means of reducing emissions should be calculated using assumptions and methodologies that are consistent with those used to calculate the above costs/tonne estimates.

The second is that only costs to meet the same objective in terms of emission reductions should be compared. If the proposed light-duty vehicle emission standards are implemented, total HC, CO, and NOx emissions from all sources in Canada would fall by approximately 12, 22 and 9 percent respectively by 1996 from the levels that would prevail if the current emission standards are maintained. It would take reductions in emissions from several sources to achieve the same level of reductions as is possible by reducing light-duty vehicle emissions. The appropriate quantity for a cost-effectiveness comparison would be the weighted average of the cost/tonne for reductions from all alternative sources needed to achieve the same aggregate emissions reductions.

4.5.2 Alternative Emission Control Strategies

Cost/tonne estimates that meet the above two conditions discussed above are not available and, for that reason, explicit, cost-effectiveness comparisons are not made. However, some cost/tonne data from United States studies presented in Tables 4.18 and 4.19 are indicative of the relative costs of alternative control strategies.

Table 4.18

Cost Effectiveness of United States Emission Control Strategies (1983 $ CDN/tonne)

Control Strategy	HC	CO
Heavy duty gasoline engine evaporative control	151	-
Heavy duty gasoline engine emission standards	135 - 467	16 - 32
Light-duty truck standards	280	18
Gasoline refueling regulations	478	-
Light-duty vehicle standards (1981)	689	60
Traffic control	903	75
Mandatory inspection and maintenance of heavy trucks	1 278	77

Source: "Revised Gaseous Emission Regulations for 1985 and Later Model Year Heavy-Duty Engines", Regulatory support document prepared by Office of Mobile Sources, Emission Control Technology Division, U.S. Environmental Protection Agency.

While direct comparisons are not possible, the relative cost-effectiveness of the proposed light-duty vehicle standards can be inferred from Table 4.18. The proposed standards are essentially the same as the 1981 U.S. light-duty vehicle standards. Table 4.18 shows that the cost/tonne to control light-duty vehicle HC and CO emissions is higher than the cost/tonne for other vehicles and gasoline refuelling regulations, but less than mandatory inspection and maintenance. However, in making the comparison, it should be noted that the aggregate reduction in emissions from the less-costly control strategies may not be as large as the reduction from light-duty vehicles.

Estimates of the cost/tonne to reduce HC, CO, and NOx from sources other than motor vehicles were recently published by the U.S. Environmental Protection Agency. These estimates have been converted to 1983 Canadian dollars and are shown in Table 4.19.

Table 4.19

Costs Per Tonne For Emission Reductions
($ 1983 CDN)

Emission/Strategy	Cost/Tonne
HC	
Stage II vapour recovery from gasoline marketing	285 - 760
Secondary seals on gasoline storage tanks	1 085
CO	
Engine modifications, catalysts, and inspection and maintenance of heavy trucks	14 - 70
NOx	
Low excess air and staged combustion at utility and industrial boilers	113 - 812

Source: "Costs and Benefits of Reducing Lead in Gasoline", draft final report, Office of Policy, Planning and Evaluation, U.S. Environmental Protection Agency, March, 1984.

The broad ranges of cost/tonne estimates for alternative emission control strategies shown in Table 4.19 preclude cost-effectiveness comparisons with the cost/tonne estimates for the proposed emission standards. However, in all cases the cost/tonne estimates for HC, CO, and NOx reductions for the proposed standards fall within the ranges shown in Table 4.19.

4.5.3 Thermal Generating Stations

Ontario Hydro Estimates

Ontario Hydro has a program to modify burners at its coal-fired generating station to reduce NOx emissions. The Nanticoke thermal generating station (TGS) was selected as the first to have its burners modified since it represents the maximum return on investment in terms of the amount of NOx removed. Ontario Hydro estimates that the cost per tonne of NOx removed at Nanticoke is about $60.

The costs per tonne to reduce NOx emissions at existing thermal generating stations vary widely. They are influenced strongly by capacity utilization factors, remaining station life, the percentage decrease in NOx emissions for the type of boilers, and the cost of the emission control equipment. For example, the capital expenditures for NOx reductions at Lakeview TGS would be similar to those for Nanticoke TGS, but, due to large differences in capacity utilization factors, expected station life and percent decreases in NOx emissions, Ontario Hydro has estimated that the cost per tonne of NOx removed at Lakeview TGS would be about $3 000.

The costs presented above are for retrofit burner modifications to reduce NOx. This technique has distinct limitations and, if greater NOx reductions are required, treatment of the outlet gas would be necessary. This approach would involve costs far in excess of the cost to modify burners. Therefore, if emissions reductions beyond those that can be obtained by modifying the burners at Nanticoke are desired, the cost per tonne of NOx removed would be much higher than $60.

Environment Canada Estimates

Environment Canada has calculated the cost per tonne to reduce NO_x emissions from existing thermal generating stations in Nova Scotia, New Brunswick, Ontario, Saskatchewan and Alberta. For all but one station the cost per tonne of NO_x removed was between $115 and $360. The estimate for the station outside the range was $2 665 per tonne (Private Communication, Bambrough, 1984). The forgoing estimates are in levelized 1982 dollars.

For construction of new plants, specifying low-NO_x burners would not result in any cost penalties.

Due to differences in the assumptions and base year used to calculate the estimates of the cost per tonne of NO_x removed, the Environment Canada estimates cannot be directly compared to the Ontario Hydro estimates.

4.6 Nonallocative Effects

Treasury Board guidelines for the evaluation of new regulations or amendments to existing regulations pertaining to health, safety, and fairness require that the evaluation includes a discussion of the nonallocative impacts of the regulations. (Treasury Board 1979). This takes the analysis beyond the confines of a benefit-cost analysis or cost-effectiveness analysis to a discussion of indirect, but not necessarily insignificant, impacts of the regulations.

Areas of analysis focus upon:

- o the distribution of costs and benefits of the proposed regulations (in effect, a review of the equity of the change)
- o the effect upon dynamic efficiency or the rate of technological change
- o the effect upon market structure or the degree of industry concentration
- o the effect upon international trade
- o the effect upon Canadian output and employment
- o the impact upon inflation.

The effect of the proposed change in the light-duty vehicle emissions standard for each of these areas is discussed in this section of the report. Where possible, the anticipated impacts are quantified.

4.6.1 Distribution Effects

In discussing the distribution effects of the proposed emission standards, there are two questions to be considered: who pays for the emission controls? and who benefits from the emission controls?

More specifically: Do some groups within society bear a disproportionate amount of the costs or enjoy a disproportionate amount of the benefits than others? Examples of the type of questions that are considered to determine if the proposed emission standards are equitable are: Do the poorest groups within society pay more or less of their income than the wealthiest groups? and, Do residents of cities benefit more or less than residents of small towns?

In order to determine the cost of the required emission control equipment to various income groups, the first step is to determine the incremental costs to consumers. This is comprised of two components--incremental capital costs (essentialy the effect of the required emission controls on the retail price of the car or the retail price equivalent of the emission control systems) and incremental operating costs or savings.

The retail price equivalent of the emission control equipment required to meet the proposed emission standards is estimated to be a maximum of $235 on average for each passenger car sold in Canada. As discussed in section 4.3.3 the maintenance and fuel economy savings of switching from leaded to unleaded gasoline are completely offset by the higher price of unleaded gasoline. Therefore, for the purpose of the analysis of distribution effects, it will be assumed that the full cost to the consumer is represented by the retail price equivalent of the required emission control equipment.

An analysis of the United States data shows that the depreciation rate is not affected by changes in the price of new cars, which means that the increased capital costs are spread over the life of the vehicle (Dorfman and Snow 1975, Harrison 1975).

To determine which income group will actually incur the costs of the emission control hardware, it is necessary to have access to data showing car ownership by income level for Canada. As these data are not available, it is necessary to make inferences from U.S. data.

The analysis of car ownership by income level data for the United States shows that the higher income groups will incur a higher level of absolute costs of the emission control equipment, since they purchase a higher number of new or relatively new cars than do lower income groups. Harrison (1975) estimates that the top four income categories will incur four times as great an absolute cost burden as the lowest three income groups. The incidence of the absolute costs of emission control equipment is expected to be similar for Canada.

However, the U.S. ownership data indicates that lower income groups pay a higher proportion of their income for the emission control systems than do the higher income groups. Harrison (1975) estimated that people in the lowest income group would pay between 0.38 and 0.42 percent of their income for the emission control system, or about twice that of those in the highest income group that pay between 0.16 and 0.20 percent of their income. It is expected that the distribution of the relative cost burden for Canada would be similar and that the distribution of the costs would therefore be regressive. That is, the lowest income groups would pay a greater portion of their income for the benefit than would the highest income groups.

The regional distribution of costs will follow the regional pattern of new automobile sales in Canada.

The regional distribution of benefits will depend upon the current air quality levels of the regions and upon the degree to which automobile emissions are a determinant of air quality levels. Discernible benefits will occur only in areas experiencing a high air pollution burden and where automobile emissions are a significant determinant of air quality.

This is likely to be the case in large cities such as Toronto, Montreal, and Vancouver, with some benefits occurring in other cities. The majority of discernible benefits will accrue to regions with high population densities. Approximately 45 percent of the Canadian population lives in the 10 largest cities. Thus, while the costs of the required emission control systems will

be borne by purchasers of cars across the country, the benefits are likely to be concentrated primarily in and around larger population centres where there is a relatively high air pollution burden and automobile emissions are a significant factor in determining air quality levels.

4.6.2 Effect on Dynamic Efficiency

This nonallocative effect is the impact of the proposed regulations upon the rate of technological change, in terms of either retarding or escalating such change.

Since the technology to meet the proposed standards is currently available for virtually all light-duty vehicles and is used for vehicles sold in the United States, there is no requirement to develop new technology in Canada. As a result, the impact of the proposed regulations on dynamic efficiency will be minimal.

While the proposed regulations may not have any impact on the rate of technological change, they may contribute to an increase in productivity using existing technology. Increased productivity may result from the decrease in the number of different emission control systems that are used on vehicles. Currently, cars produced in Canadian and U.S. factories are equipped with at least two types of emission control systems--one for Canada and one for the United States--on the same assembly lines. The proposed Canadian standards would eliminate the production of vehicles with emission systems designed specifically for the Canadian market. This may lead to increased production efficiencies and reduced inventory requirements.

4.6.3 Effect on Market Structure

There are two separate issues to be considered in this area: the effect the proposed regulations may have upon industry concentration, and the degree to which the regulations may serve as an effective barrier to market entry.

With respect to the level of industry concentration, it is unlikely that the proposed regulations will lead to a higher level of concentration in what is already a highly concentrated industry.

For the most part, automobile manufacturers produce and market vehicles worldwide. Given the importance of the United States automobile market, most car manufacturers currently manufacture a line of vehicles certified to meet U.S. emission standards that will also comply with the proposed Canadian standards. For these manufacturers, the change in standards does not raise any significant entry barrier to the Canadian market.

There are a small number of importers currently active in the Canadian market that do not manufacture vehicles complying with current U.S. emission standards. To these manufacturers the costs of developing and certifying emission control systems to meet the proposed Canadian emission standards may prove to be an effective barrier to the Canadian market. Since these costs have to be recovered on relatively low sales volumes in Canada, the proposed emission standards may be an effective barrier to entry.

The light-duty vehicles affected are: Suzuki, Lada, Pony, Dacia, Skoda, and Innocenti. The Canadian distributors of the three highest selling models--Suzuki, Lada, and Pony--were contacted for their comments. The distributor for Lada cars was the only one to reply to the request for comments. Lada Cars of Canada Inc. estimated that the cost of meeting the proposed emission standards would be approximately $700 per vehicle. The Canadian distributor would be responsible for purchasing and fitting the emission control hardware. The costs would have to be passed through to the retail level. This would cause a large decrease in Lada's price advantage. As a result, Lada Cars of Canada Inc. estimates it would lose 50 percent of its sales, or 3 500-4 000 units per year.

Another issue that should be considered here is the effect of the proposed regulations on the relative competitive position of the various manufacturers in the Canadian automobile market. A survey of automobile manufacturers to determine the cost of complying with the proposed emission standards (DeKany and Piloruso 1983) showed that the cost of compliance may

be higher for offshore manufacturers than for North American manufacturers. Ford and Chrysler indicated that the retail price equivalent of the emission control system required to comply with the proposed standards would be $175 and $210, respectively. General Motors declined to provide cost or price estimates. Retail price equivalents of the required emission control systems provided by Honda, Toyota, and Renault ranged from $345 to $805, $373 to $452, and $650 to $940, respectively.

Based upon the submitted cost data, offshore manufacturers are expecting to incur a higher cost of compliance than their North American counterparts. If this apparent cost penalty resulted in a proportionately higher increase in the retail price of imported cars than domestic cars, a shift in market share from imported to domestic cars would be expected. However, at least in the case of the Japanese imports, which accounted for more than 70 percent of all Canadian imports during 1983, a loss of market share is unlikely to occur. The Japanese importers have traditionally set their prices under those established by the domestic manufacturers. The Japanese motor vehicle manufacturers obtain higher profit margins in the North American market than domestic manufacturers due to their lower production costs. This allows the Japanese considerable flexibility in establishing retail prices. For these reasons the Japanese motor vehicle manufacturers are unlikely to increase retail prices by an amount greater than the domestic manufacturers regardless of their costs of compliance.

Even if the retail price of imported Japanese cars increased by a greater amount than domestic cars, sales volumes would be unlikely to fall. The number of Japanese cars allowed into Canada has been limited by negotiated quotas since 1981 and, as a result, the number sold in Canada is limited by supply. As long as some form of restraints on imports of Japanese cars remain in place, demand for them is likely to exceed supply. Under these circumstances price increases of the magnitude implied by the apparently higher costs of compliance would have little or no effect on sales volume. Indeed, the average price of Japanese cars sold in Canada has been increasing by a greater amount than the average price of domestic cars since the quotas have been in place (Statistics Canada 1984d). This has had no effect on sales volume.

Imports of cars from countries other than Japan are not restrained by quotas. In cases where the costs of compliance with the proposed emission standards results in a larger retail price increase for imported cars than for domestic cars some sales of imported cars would be replaced by sales of domestic cars. Therefore, implementation of the proposed emission standards could result in a slight shift in market share in favour of domestic manufacturers.

4.6.4 Effect on International Trade

Treasury Board guidelines indicate that there are three areas that should be considered with respect to international trade: the effect upon the balance of payments, any effect upon the international competitiveness of Canadian products, and whether or not the regulations act as a nontariff barrier to trade.

There should be no effect upon the international competitiveness of Canadian products, as the regulations will apply only to light-duty vehicles sold in Canada. Manufacturers are not compelled to apply the standards to export products, and so remain free to design and produce products specifically for export markets, as is the current practice.

The proposed standards will apply to all automobiles sold in Canada and, as such, do not discriminate between domestic and imported products. As a result they are technically not a nontariff barrier to trade. However, the proposed standards do place an extra burden on small manufacturers, as discussed in the previous section. The small manufacturers affected are offshore, and the proposed standards may affect their ability to compete in Canada.

The required emission control systems will be imported into Canada in two ways: as individual components for vehicles assembled in Canada for sale in Canada, and on fully-assembled vehicles imported into Canada. The value of components imported for final assembly in Canada will be lower than the value of emission controls on imported vehicles since the cost of these components will not include the vehicle assemblers' markup.

For 1980 and 1981, an average of 18 percent of the automobiles sold in Canada were manufactured in Canada (Automotive News 1982 Market Data Book). Almost all of the Canadian production that was exported went to the United States. Of the other cars sold in Canada, 54 percent were manufactured in the United States; and the remaining 28 percent were imported from offshore. Cars produced in the United States for sale in Canada represent about 6 percent of total U.S. production. It is assumed that these percentages will not change significantly during 1987 to 1996.

The cost of the required emission control equipment is estimated (from the disaggregated cost data presented in DeKany and Pilorusso 1983) to be a maximum of $143, on average, per car, to the vehicle manufacturer. Of this total cost, it is estimated that $136, or 95 percent, will be sourced outside of Canada. A survey of manufacturers assembling passenger cars in Canada revealed that virtually no emissions control hardware is sourced in Canada. A small volume of parts, e.g., metal stampings for catalytic converters, is produced in Canada, shipped to U. S. for assembly into components that are, in turn, shipped to U. S. and Canadian vehicle assembly plants.

The value for duty of the required emission control equipment for vehicles produced outside of Canada is estimated to be $173. This figure includes the vehicle assemblers' markup on the cost of the emission control equipment.

As indicated in section 4.1.1 of this report, it is estimated that 85 percent of the passenger cars sold in Canada will be affected by the proposed regulations. It has been assumed here that cars produced in Canada and cars imported to Canada are affected in equal proportions by the proposed regulations.

In Table 4.20 the estimated increase in the value of imports as a result of the proposed regulations are shown. The estimates are based upon the sales projections developed in section 4.1.1 and the assumptions stated above.

Table 4.20

Estimated Increase In The Value Of Imports As a Result of The Proposed Regulations (millions of 1983 $)

Year	For Automobiles Assembled in Canada	For Imported Automobiles	Total
1987	$ 20	$ 119	$ 139
1988	20	119	139
1989	20	120	140
1990	21	121	142
1991	21	121	142
1992	21	122	143
1993	19	110	129
1994	17	99	116
1995	19	110	129
1996	21	122	143

To put the estimated increase in the value of imports into perspective, imports of automotive products from the United States were $17.6 billion in 1983 (Statistics Canada 1984b). Canada's surplus in trade in automotive products was $3.3 billion in 1983. Thus, the highest estimate of increased imports resulting from the proposed regulation (i.e., during 1992) would have increased total imports in 1983 by about 0.8 percent and decreased the trade surplus by 4.3 percent.

Canada's current account balance of trade in 1982 was a surplus of $3 017 million; in 1981 the current account showed a deficit of $5 766 million and in 1980 a deficit of $1 069 million (Bank of Canada Review 1983). The highest forecast level of imports (1992) would have decreased the 1982 surplus by 4.7 percent, and increased the 1981 and 1980 deficits by 2.5 and 13.4 percent, respectively.

Another issue to be considered with respect to international trade is the effect of the proposed emission standards on Canada's energy consumption and the consequence on energy trade. As discussed in section 3.3.5, the proposed standards are not expected to have any impact on fuel consumption for about 85 percent of the light-duty vehicle fleet during 1987-1996. Therefore, there is no anticipated impact on energy consumption and trade.

The final issue related to international trade is the impact that implementing the proposed emission standards would have on the number of U.S. citizens that purchase cars in Canada. Under the terms of the Canada-U.S. Automotive Products Trade Agreement, U.S. citizens can purchase a car in Canada and import it to the United States, duty-free, as long as it meets U.S. emission and safety standards. The U.S. buyers would also be eligible for a refund of the federal and provincial sales taxes. The number of cars imported to the United States from Canada in this way is relatively small at this time since most models sold in Canada do not meet U.S. emission standards and there is a general lack of knowledge on the part of the U.S. car buyers. Canadian and U.S. safety standards are substantially identical. If the proposed emission standards are

implemented, Canadian and U.S. emission standards will also be identical, making virtually all cars sold in Canada eligible for private import to the United States.

Cars sold in Canada are currently less expensive than comparable models sold in the United States. Canada/United States price comparisons are shown in Table 4.21. The Canadian prices are net of the federal sales tax, and U.S. prices have been converted to 1983 Canadian dollars to make the columns of Table 4.21 directly comparable. Prices for models with manufacturers' suggested retail prices less than $15 000 are 9 to 10 percent lower, on average, in Canada than in the United States.

It was possible to separate out some of the cars sold in Canada that are equipped with emission control systems that will meet the proposed regulation (i.e., U.S. federal systems). They are: G.M. (Feedback Fuel System), Renault (U.S. built), and most, but not all, cars selling for more than $15 000. Models in this category generally sell for about 5 percent less in Canada than in the United States. With the exception of U.S.-built Renaults, the U.S.-specification cars sold in Canada are generally at the high end of the model lines and/or are sold in limited quantities.

Table 4.21

Average Retail Prices For 1984 Model Cars
(1983 Canadian $)

	Retail Price Less than $15 000			Retail Price More than $15 000		
	CDA	U.S.	U.S. as % of CDA	CDA	U.S.	U.S. as % of CDA
General Motors						
Buick	10 409	11 326	109	19 198	20 126	105
Cadillac				26 000	26 528	102
Chevrolet	8 673	9 453	109	28 345	28 966	102
Oldsmobile	10 146	11 184	110	17 305	18 424	106
Pontiac	9 577	10 611	111	17 279	17 902	104
G.M. (Feedback Fuel System)	11 098	11 613	105			
Ford						
Ford	9 431	10 420	110	16 540	17 475	106
Lincoln				28 118	27 859	99
Mercury	10 040	10 974	109			
Chrysler						
Chrysler	11 372	12 445	109	15 558	17 204	111
Dodge	8 610	9 696	113			
Plymouth	7 654	8 682	113			
American Motors						
Renault (all)	8 722	9 308	107			
Renault (U.S. built)	7 685	8 042	105			
Eagle	11 535	12 540	109			

Sources: Canadian Prices - 1984 Motor Vehicle Data Book, Vol. 34
Sanford Evans Communications Ltd.
U.S. Prices - AUTO '84, January 1984, Consumer Guide

If the Canada/United States price differential is maintained and the proposed emission standards are implemented, the number of U.S. citizens buying cars in Canada can be expected to increase. The recent devaluation of the Canadian dollar has increased the price incentive, and advertising by Canadian dealers in U.S. border towns has raised the awareness of the cost savings available to U.S. citizens (Automotive News 1984). Implementing the proposed emission standards would make available in Canada almost as wide a choice of makes and models to U.S. citizens as they have in their own country.

If the current price differential between Canada and the United States is maintained after the proposed emission standards are implemented, car dealers in Canadian border towns can be expected to increase their sales to U.S. citizens. Since these cars would be exported there would be a positive impact on Canada's balance of trade.

In the absence of administrative or regulatory measures to prohibit private importation of cars by U.S. citizens, the domestic motor vehicle manufacturers can be expected to come under intense pressure, from dealers in U.S. border towns, to increase car prices in Canada to levels that would discourage U.S. citizens from purchasing cars in Canada.

4.6.5 Employment and Output Effects

Companies that are to be directly affected by the proposed regulations--i.e., automobile assemblers and muffler manufacturers--were contacted to determine the extent of the effect of the proposed regulations on employment.

General Motors indicated that its Canadian employment would decline by the 16 to 20 employees who are currently responsible for developing and certifying unique emission control systems to meet existing Canadian emission standards. All of the other companies contacted indicated that there would be no meaningful change in employment caused by the proposed standards.

In addition to direct changes in employment, it is necessary to consider the secondary effects caused by the higher price of automobiles equipped with the required emission control systems.

Section 4.6.4 on the balance of payments implications provided estimates of the increase in imports required for compliance with the proposed regulations over the period 1987-1996. This outflow of funds will be equal to the sum of reduced expenditures on other items and reduced savings by car buyers. As a result of these reduced expenditures labour incomes and gross provincial product will decline and fewer jobs will be created in Canada.

Estimates of the magnitude of these effects have been made with the use of Statistics Canada's Interprovincial Input-Output Model (Gaston and Bedard 1980). In the input-output model, labour income is defined as the sum of wages and salaries, supplementary labour income, and unincorporated business profits. Gross provincial product is equal to labour income plus corporate operating surplus, and employment is measured in person years.

It is emphasized that these estimates of the economic effects of the proposed automobile emission regulations are based on the structure of the economy as it was in 1974. (An updated version of the interprovincial input-output model is in preparation). Changes since 1974, especially in labour productivity and Canada's dependency on imports, suggest that the quantitative magnitude of these economic impacts could be significantly less than the estimates reported below. This may be compensated for in part by the omission in the input-output model of any repercussions of an increase in imports on reduced corporate profits and tax revenues, both of which could add to the severity of the estimated impacts. However, the input-output model also omits the possibility of adjustments in Canada's balance of payments in response to an increase in imports. Although these adjustments are difficult to estimate it is likely that they would mitigate the adverse impacts of the initial rise in imports. Furthermore, the impacts were estimated on the assumption that the

reduction in expenditures by Canadians would be divided between Canadian made goods and services and imports according to the average pattern of consumption prevailing in 1974. If, at the margin, imports supply a greater proportion of Canadian consumption than the 1974 average, then the impacts would be over estimated. However, in the contrary case, where the marginal propensity to import is less than the average, the impacts would be underestimated.

Finally, the possibility discussed in section 4.6.3, that there could be some degree of substitution of Canadian made cars for offshore imports as a result of the proposed regulations is a final reason for regarding the impact estimates as probably too high.

The results of the input-output calculations suggest that the proposed emission standards could lead to a reduction in Canadian labour income of up to $80 million per year, a reduction in Canadian GNP of up to $120 million per year, and possibly, as many as 3 200 fewer jobs created in Canada each year than would be the case if the money spent on additional imports of emission control hardware was spent in Canada.

To put these figures in perspective, labour income in 1982 was $208.2 billion (Statistics Canada 1983a). The projected drop in labour income represents no more than 0.04 percent of labour income in 1982. Canadian GNP in 1982 was $356.2 billion (Statistics Canada 1983b). The projected drop in GNP represents about 0.03 percent of GNP in 1982. The number of people employed in Canada grew by 90 000 in 1983 over the year earlier after falling by 362 000 in 1982 (Statistics Canada, 1984c). The upper limit of 3 200 jobs foregone represents 3.6 percent of the number of net jobs created during 1983 and 0.9 percent of the net jobs lost during 1982.

The provincial distribution of these effects would probably follow the general pattern of economic activity across the country, with most of the impacts falling in Ontario, Quebec, Alberta, and British Columbia. Of course, these provinces are also in a better position to absorb these effects owing to the greater size and diversity of their economies.

4.6.6 Effect Upon Inflation

The question of the effect of the proposed regulations on inflation is relatively complex. If it is assumed that the costs of complying with the proposed regulations will be passed through to the consumer, there is no question that there will be increased automobile prices caused by the proposed regulations. This price increase has been estimated to be $235, on average.

While the magnitude of the price change is clear, the inflationary impact is not. Statistics Canada would not include the price change in the official calculation of the Consumer Price Index (CPI). This reflects the policy of Statistics Canada that product changes resulting from legislated changes to emission standards are taken to be quality improvements and, as a result, are discounted from pure price changes (Private Communication, DeVries 1984).

If the increase in retail price is assumed to be a pure price change the impact on inflation is as follows. Statistics Canada (1984d) reported that the average price of a car sold in Canada in 1983, was $10 719. If the full retail price equivalent for emission control hardware of a maximum of $235 is passed through to the consumer, car prices will increase, on average, by 2.2 percent. This is about the same amount as average car prices have increased in each of the past two years. In calculating the consumer price index (CPI), Statistics Canada weighs the new automobile purchase component of the index as 5.42 percent of the total. Therefore, if the price increase caused by the new emission systems was to be included in the calculation of the CPI, it would result in a 0.1 percent increase in the CPI.

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